Monitoring, Modelling and Health Impacts of Air Pollutants Arising from the Maptaphut Industrial Estates, Thailand

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Abstract

The Maptaphut Industrial Estate is located on the Gulf of Thailand, Rayong Province. The area, which has been designated as a main centre for the petrochemical industry currently occupies 16 sq km and comprises petrochemical plants, chemical and fertilizer plants, refineries, construction plants, and steel industry; there are also residential and commercial areas (IEAT, 2004). There is a significant population around the site, with 24,000 inhabitants in the immediate vicinity according to Jadsri *et al* (2006). The estate has been held responsible for deaths and hospital admissions due to leaks and accidents dating back as far as 1997. Whilst the environmental and health and safety performance of the estate as a whole has significantly improved over recent years, there are still significant outpatient admission rates to Maptaphut hospital for respiratory illness, as recently reported by Jadsri *et al.* (2006), raising the question of whether local emissions are significantly contributing to ill health, or whether general background concentrations of pollutants from nearby road sources and from Rayong City are the main contributions. The main aim of this research, therefore, was to accurately model the dispersion of pollutants from the estate, and to attempt to quantify the health impacts of these emissions.

The specific objectives of this study were to (a) to characterise meteorological conditions in the Maptaphut area; (b) to develop a multiple linear regression statistical model to characterise and predict atmospheric pollutant concentrations in Maptaphut; (c) to investigate the relationship between air pollution and ill health in Maptaphut using a multiple linear regression statistical model; (d) to evaluate the effectiveness of Gaussian and Computational Fluid Dynamics atmospheric dispersion modelling software packages in predicting ground level pollutant concentrations at points around the industrial estate and (e) to use the results of the dispersion modelling studies to assess the contribution of the industrial estate to the overall atmospheric pollutant load in the Maptaphut area, and from published health impact factors, to assess the overall health impact of the estate.

The first objective was to characterise the environmental status, trend, and impacts of air pollution during the period 1998 to 2007. The estate is located in the coastal area; thus, the role of the sea-land breeze has a significant role in the dispersion of air pollutants harmfulness. Data collected for the Maptaphut Industrial Estates area, including regional, temporal and spatial considerations included: meteorological data from 100-metres tall meteorological mast; ambient air quality data from three ambient air quality monitoring stations; industrial emissions data; traffic volume on nearby major

roads; and outpatient admissions data at the Maptaphut and Rayong hospitals. Comparisons with the ambient air quality in the Bangkok area were made, and the daily and yearly trends in concentrations of the main air pollutants were analysed.

Multiple linear regression models correlating pollutant concentrations with respiratory outpatient admissions rates showed that O_3, PM_{10} and NO were statistically significant determinants. The overall correlation had a coefficient of Determination (R^2) of 41.4% for one week average data, increasing to 51.2% when air temperature and %RH were included. Accumulation effect of pollutants up to four weeks period exposure does not appear to have an effect. A basic health impact analysis study using the ADMS modelled concentrations and the WHO AirQ tool, along with default risk factors, showed that emissions from the Maptaphut industrial estate account for almost all of the NO_2 and SO_2 related respiratory illness and between 10 and 27% of the PM_{10} related admissions; this actually represents less than 2% of the total respiratory admissions for this area.

Furthermore, statistic models were developed to predict daily maximum of 1-hour Ozone and PM_{10} concentration by multiple linear regression based on 1998-2002 statistical data. Average percentage errors for the model then applied to the prediction of time series were 21% and 38% for Ozone and PM_{10}, respectively. The effectiveness of a range of dispersion models in predicting ambient pollutant concentrations from the industrial estate was also investigated. Of these, ADMS showed the best performance; it was able to predict sulphur dioxide concentrations with a reasonable degree of accuracy and both the magnitude of the peaks and the general trends were reasonably characterised. This indicated that the meteorology model used in ADMS, and the treatment of the boundary layer and Monin-Obukhov lengths is applicable to South East Asian climatic conditions.

For PM_{10} and NO_x, where there were significant other sources apart from the industrial estate, the characterisation was less good, with some significant under predictions of PM10 in particular. The predicative capability of ISC for SO_2, NO_x and PM_{10} was generally very poor. There was some coincidence of peaks, but generally the peaks were over predicted for SO_2 and NO_2 and under predicted for PM_{10}. Finally, for PanEIA, which is a this computational fluid dynamics (CFD) dispersion model, the model was able to predict general trends and peaks, particularly for SO_2; however there was a tendency to significantly overestimate the daily average concentrations. In addition, run times are significantly greater than those for either ADMS or ISC, possibly by a factor of 10.

On the basis of the research findings, the following recommendations were made to the regulatory authorities: (a) that ADMS be adopted for regulatory purposes in plae of ISC3, and that other advanced Gaussian models such as AEROMOD be evaluated; (b) that further research be carried out on statistical approaches to predicting PM_{10} and ozone concentrations in conjunction with specific data sets collected from mobile meteorological and air quality monitoring stations; (c) in view of the conclusion that that ambient air quality in populated areas adjacent to the Maptaphut estate is not adequately characterised by the locations of the current monitoring stations it is recommended that either additional monitoring stations are installed in areas of high pollution levels or that use be made of mobile air quality monitoring stations are used routinely to collect air quality data at several points around the industrial estate

Further work should include: (a) the compilation of a more accurate pollutant inventory in addition to the industrial emissions from Maptaphut; (b) the further investigation of chemistry and coastal modules of ADMS in order to build a more accurate model, including atmospheric chemistry; (c) further work to be carried out on statistical techniques for predicting ambient pollutant concentrations, preferably in conjuction with additional monitoring data from mobile monitoring; and (d) repeat the health impact analysis studies based on more complete hospital admission data including: age, sex, district of residence.

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Declaration

I declare that the work contained in this thesis has not been submitted for any other award and that it is all my own work.

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Contents

Chapter 1

Introduction

1.1 Background

The detrimental health effects of high levels of air pollution are well known. A recent project, APHENA, has attempted to bring together European and North American short-term epidemiological studies (Samet, Katsouyanni & Principal Investigators, 2006). The project has recently reported on the effects of PM_{10} which showed that for all causes of mortality, across all ages and cities a range of 0.2% to 0.6% increase in daily mortality is observed for a 10μg m^{-3} increase in PM_{10} concentration (Samoli *et al.*, 2008). Similarly in Asia, a recent project, PAPA, has been set up with the aim of meta-analysing short term epidemiological studies published for different Asian cities (Wong et al., 2008). The study has recently published a four-city meta-analysis that has revealed health impacts to be as high (PM_{10}) or higher (NO_2 and SO_2) than observed in the Western studies. These studies are discussed further in Chapter 2.

In many parts of the world there have been significant improvements in air quality in recent times, often initially in the form of reactive legislation such as the 1956 and 1968 Clean Air Acts following the London smog events, but then followed up by more coordinated, holistic, regimes, which are usually now based on the recommendations of the World Health Organisation Guidelines (WHO 2005). In Thailand, for example, there are the National Primary Ambient Air Quality Standards (NPAAQS) that form part of the ambient air pollution control regime that is regulated by the Department of Pollution Control in the Ministry of National Resources and Environment, (Wangwongwatana, 2001).

Nevertheless, despite the undoubted improvements in air quality in some parts of the world there are still some alarming statistics; for example the recent UN Global Environmental Outlook (GEO4) Report 2007 which stated that "Indoor and outdoor air pollution is responsible for more than 2 million premature deaths annually. Poor women are particularly vulnerable to respiratory infections, as they have high levels of exposure to indoor air pollution." In Europe, the European Union's Thematic Strategy on Air Pollution, 2005, stated that "...recent studies suggest that about 370,000 Europeans die prematurely each year as a result of air pollution – more than 90% of these premature deaths are

caused by fine particulates and most of the remainder by ground level ozone." (Murley, 2007). There is growing evidence that even very slightly elevated levels of air pollution can produce in significant health effects, shortening life spans and exacerbating existing medical complaints (WHO 2005).

In Thailand, the location for this study, there are significant problems with particulate matter and carbon monoxide, principally due to traffic pollution. In Bangkok for example, the congested traffic causes start-stop cycles of vehicles with low speed: data for the year 2000 shows that approximately 28% of 24-hour average concentrations of roadside ambient TSP exceeded the ambient air quality standards.

As well as traffic related pollution, there are also problems pertaining to industrial emissions in Thailand, and it is this aspect that provides the basis for the research carried out in this thesis. The country is an attractive location for foreign companies to site their industrial operations; this is principally because of the abundance of natural resources, relatively cheap labour, and convenient transportation. Investment of this kind is actively supported by the Thai government through the Industrial Estate Authority of Thailand, and has resulted in the development of industrial estates in many parts of the country, especially in the central and eastern parts (IEAT, 2004).

The particular area of interest for this study is the Maptaphut Industrial Estate in the East of the country, on the Gulf of Thailand, Rayong Province (about 220km from Bangkok). This area has been was designated by the government as a future home for Thailand's petrochemical and heavy industries (IEAT, 2004). The total area for the site is 16 sq km (10,215 Rai), comprising petrochemical plants, chemical and fertilizer plants, refineries, construction plants, and steel industry; there are also residential and commercial areas (IEAT, 2004). There is a significant population around the site, with 24,000 inhabitants in the immediate vicinity according to Jadsri *et al* (2006).

The estate has had a chequered history in terms of environmental protection: in 1996, people were hospitalised after being poisoned by phenol and sulphuric acid; in the same year, illegally-stored oil drums exploded, killing 17 people and contaminating ground water; in 1997, 40 school children were admitted to a hospital due to nausea, vomiting and chest pains, a day after complaining of strange odours (sulphur dioxide); in 1997, the Harbour Department found higher than standard mercury levels in wells; in 2000, about 200 residents of Maptaphut were hospitalised after carbonyl chloride leak (Campaign for

Alternative Industry Network, 2003). In 2005, Greenpeace Southeast Asia showed evidence that multinationals at the Maptaphut Industrial Estate were releasing toxic chemicals into the atmosphere, including benzene, vinyl Chloride, 1,2-dichloroethane (EDC), chloroform and 20 other toxic substances (Greenpeace, 2005). Recently under the regulation of the, environmental standards have improved with companies encouraged to seek accreditation to the ISO 14001 Environmental Management standard. It is worth noting that there is a separation of regulatory powers with respect to Maptaphut, and indeed all of the industrial estates: emissions from the processes in the estates are regulated by the Industrial Estates Authority of Thailand (IEAT) whereas ambient air quality is monitored and regulated by the Pollution Control Department. The IEAT could be viewed as having a conflict of interest, i.e. wishing to maintain momentum in terms of investment yet at the same time acting as a regulator of environmental emissions.

Despite recent improvements in environmental performance by the estate as a whole, there are still significant outpatient admission rates to Maptaphut hospital for respiratory illness, as recently reported by Jadsri *et al.* (2006), raising the question of whether local emissions are significantly contributing to ill health, or whether general background concentrations of pollutants from nearby road sources and from Rayong City are the main contributions. The main objectives of this research, therefore, are to accurately model the dispersion of pollutants from the estate, and to attempt to quantify the health impacts of these emissions.

1.2 Objectives

1) To characterise meteorological conditions in the Maptaphut area.
2) To develop a multiple linear regression statistical model to characterise and predict atmospheric pollutant concentrations in the Maptaphut, Thailand.
3) To investigate the relationship between air pollution and ill health in the Maptaphut, Thailand using multiple linear regression statistical model.
4) To evaluate the effectiveness of Gaussian and Computational Fluid Dynamics atmospheric dispersion modelling software packages in predicting ground level pollutant concentrations at points around the industrial estate.
5) To use the results of the dispersion modelling studies to assess the contribution of the industrial estate to the overall atmospheric pollutant load in the Maptaphut area, and from published health impact factors, to assess the overall health impact of the estate.

1.3 Overview

Chapter 2 reviews the ambient air quality and meteorological monitoring regime in Thailand generally, and also specifically in the Maptaphut area; the instrumentation and underlying physical and chemical principles of operation are described. There is an extensive review of published health effects of all the major atmospheric pollutants. Finally, statistical and dispersion modelling techniques for predicting pollutant concentrations are discussed.

Chapter 3 discusses the methodology used in this study, specifically: the analysis of meteorological and pollution data in the Maptaphut area; the development of a multiple linear regression (MLR) statistical model using SPSS program to predict ground level pollutant concentrations; the use of MLR techniques to investigate the relationship between ground level pollutant concentrations and hospital respiratory admission rates; and the evaluation of the effectiveness of various dispersion modelling packages in predicting ground level concentrations of major pollutants released from the industrial estate (two Gaussian packages, ADMS and ISC, together with a Computational Fluid Dynamics package, PanEIA).

Chapters 4 and 5 present the results for these studies and discuss their significance in terms of likely health effects attributable to the industrial estate.

Chapter 2

Background and Literature Review

2.1 Air Pollution in Thailand

Air pollution is a local, regional and global problem. At a local level particulate and gaseous pollutants (CO, CO_2, SO_2, NO_x, O_3) affect human health and may create other problems such as low visibility; however, these gases and small particles can also be transported over large distances, contributing to regional and global problems such as greenhouse effect, global warming, climate change and ozone depletion. In Thailand, the major air pollution problems are the particulate matter and carbon monoxide that are emitted from industrial sources and vehicles in urban area, and open burning in agricultural areas (Poboon, 2006 and PCD 2006). The Pollution Control Department is the government sector under the Ministry of Natural Resources and Environment that develops environmental quality management plans to control, prevent, and mitigate environmental problems. As part of this process, emission standards are established for pollution control from point sources in order to meet the ambient environmental quality standards. The standards are in accordance with the Enhancement and Conversation of the National Environment Quality Act B.E. 2535 (1992) and are the national minimum standards for ambient air quality. As part of the strategy to ensure compliance with these standards, a number of ambient air quality and meteorological monitoring networks have been developed to monitor air quality status in Thailand.

The following section reviews the instrumentation, data acquisition and remote data retrieval used at the ambient air quality monitoring stations used by the Thai Pollution Control Department.

2.1.1 Review of Thailand's Ambient Air Quality and Meteorological Monitoring Networks

In Thailand, ambient air quality monitoring has been promulgated by NPAAQS (National Primary Ambient Air Quality Standards) since 1981. At first, the air quality in Bangkok and its surrounding vicinity was mainly monitored. Since then, the network has been improved, upgraded, and expanded to cover other areas of the country.

In 1983, the first air quality monitoring system was installed. It consisted of eight permanent ambient air quality monitoring stations located around Bangkok. The stations were located in residential, commercial, industrial, and mixed areas. These stations were operated continuously; however they were manual systems, rather than the on-line systems used now, requiring a large number of man-hours to collect, transfer, and analyse the data.

In 1987, the second air quality monitoring system, which was the first on-line and real-time continuous monitoring system in Thailand, was installed with the assistance from the Japan International Cooperation Agency (JICA). There are five automated air quality monitoring stations located in Samut Prakarn province, an industrial area southeast of Bangkok. The collected air quality monitoring data from each monitoring station was transmitted via dial-up telephone line to a central processing computer for data storage and analysis.

In 1991, the third monitoring system, also an on-line and real-time continuous system, was installed. The system consisted of four monitoring stations located at roadside locations on the streets of Bangkok. The public could know the situation of the air quality in that area by the display board of each station; at the same time the data was simultaneously transmitted via a dedicated telephone line to a central processing computer for data storage and analysis. Additionally, in approximately 22 of the most congested streets in Bangkok, short-term temporary roadside air quality monitoring was also undertaken for a two-week period every year in each street (Wangwongwatana, 2001).

In 1992, a nation-wide ambient air quality monitoring network and a meteorological monitoring network was designed by the Pollution Control Department (PCD), with technical assistance from the Swedish Government. The networks were gradually installed over several phases through upgrading of the existing air quality monitoring stations and installing new ones.

2.1.2 Current Ambient Air Quality Monitoring Network

At present, there are 52 automated ambient air quality monitoring stations located throughout the country as detailed in Table 2.1. The ambient air pollutants monitored in these stations are: CO, NO_x, SO_2, O_3, TSP, PM_{10}, Pb, HC, and H_2S. Meteorological data is

also monitored via 10-metre or 30-metre meteorological masts that are fitted in forty five of the stations to measure wind speed, wind direction, temperature, humidity, barometric pressure, solar radiation, and precipitation. In addition, 100-metre tall meteorological masts were located in five provinces to collect meteorological data from all main part of Thailand.

All monitoring stations in Thailand are operated automatically with remote control from the central computers located at the PCD in Bangkok. In order to ensure that good data quality are acquired from the monitoring stations, automatic zero and span calibrations of gas analysers in the stations are performed every day. Additionally, manual calibrations with standard gases complying with of US Environmental Protection Agency (USEPA) protocol grade are carried out every fifteen days.

Ambient air quality monitoring data is collected on an hourly basis, then subsequently transmitted daily to the central data processing system at the PCD through a dial-up telemetric communication system and then further analysed by specialists.

With respect to the data transmission system, there are five regional node computers. Regional node computers are located in Bangkok for the central region, Chiangmai for the northern region, Khon Kaen for the north-eastern region, Chonburi for the eastern region, and Song Khla for the southern region. Regional node computers retrieve air quality and meteorological data via dial-up telephone lines from the monitoring stations located in their respective regions on a daily basis. The central node computer serves not only as a regional node computer, but also the central processing computer; this node connects the other four regional node computers daily to retrieve data, which are stored in the database for further processing and reporting. In addition, the PCD has five mobile ambient air quality monitoring units for emergency response to air pollution episodes and other special air pollution studies.

Table 2.1 Spatial Distribution of Air Quality Monitoring Stations and 100-metre Tall Meteorological Masts

Region	Province	Number of Air Quality Monitoring Stations	Number of 100-metre Meteorological Masts
Central Region (34 stations)	Bangkok	18	1
	Samut Prakarn	5	-
	Patum Thani	1	-
	Nonthaburi	2	-
	Nakorn Pathom	1	-
	Ayudthaya	1	-
	Saraburi	2	-
	Ratchburi	1	-
	Samut Sakorn	2	-
Northern Region (8 stations)	Chiangmai	2	1
	Lampang	4	-
	Nakorn Sawan	1	-
North-eastern Region (3 stations)	Khon Kean	1	1
	Nakorn Rachsima	1	-
Eastern Region (8 stations)	Chonburi	3	-
	Rayong	4 (29T, 30T, 31T inclusive)	1
Southern Region (4 stations)	Surat Thani	1	-
	Phuket	1	-
	Songkhla	1	1
Total		**52**	**5**

Source: PCD Thailand

2.1.3 Current Meteorological Monitoring Network

In addition to the 10-metre meteorological masts at most of the air quality monitoring stations, the PCD has a dedicated meteorological monitoring network consisting of five 100-metre meteorological masts located in Bangkok for the central region, Chiang Mai for the northern region, Khon Kaen for the north-eastern region, Rayong for the eastern region, and Song Khla for the southern region and one Radio Acoustic Sounding System (RASS) for monitoring upper-air meteorological condition.

Meteorological parameters are measured at different heights on each 100-metre meteorological mast as indicated in Table 2.2. Wind speed, 3-dimensional wind directions, temperature, and relative humidity are measured at three different heights, namely 10, 50 and 100-metres. Solar radiation, net radiation, barometric pressure, and precipitation are measured at a height of two metres above the ground.

Table 2.2 Parameters measured on a 100-metre meteorological mast at different heights

Height Above Ground (m)	Parameters Measured
2	Solar radiation, Net radiation, Barometric pressure, and Precipitation
10	Wind speed, 3-dimensional wind direction, Temperature, and Relative humidity
50	Wind speed, 3-dimensional wind direction, Temperature, and Relative humidity
100	Wind speed, 3-dimensional wind direction, Temperature, and Relative humidity

Source: PCD Thailand

2.1.4 Ambient Air Quality and Meteorological Monitoring Network in Rayong Province

The meteorological and ambient air quality data used in this study was provided by three ambient air quality monitoring stations and one 100-metre meteorological mast tower in Rayong province, belonging to the Thai Pollution Control Department. In this area, there

are three industrial estates: Maptaphut Industrial Estate, Eastern Industrial Estate and Padaeng Industrial Estate all of these are located in Amphoe Ban Chang, along the coastal line between the Gulf of Thailand and Road 3 (Figure 2.1). The locations of the monitoring stations are strategically placed to cover the dispersion of air pollution from these industrial estates.

Locations of the ambient air monitoring stations are indicated in Figure 2.1. Stations 29T and 31T are located to the North of the industrial estates and around four kilometres apart. Both of them are down stream of sea breeze, which passes the industries. The 29T station located in residential area and the 31T station is located in an open land space and about 200 metres distance from a heavy traffic highway, number 3. Although there is a 30-metre meteorological tower attached to this station, nearby to this station there is one of the 100-metre meteorological towers. Station 30T is located in the highly populated Rayong town to the east of Maptaphut, about 15 km distance from the industrial estates. The distance from the stations to the Gulf of Thailand is about two, four and six kilometres for 30T, 29T and 31T, respectively. With regard to the roads, both road 3 and road 36 are busy, with traffic density in 2001 of 28,375 vehicles/day and 25,447 vehicles/day, respectively.

Photographs of the monitoring stations are shown in Figures 2.2 to 2.12.

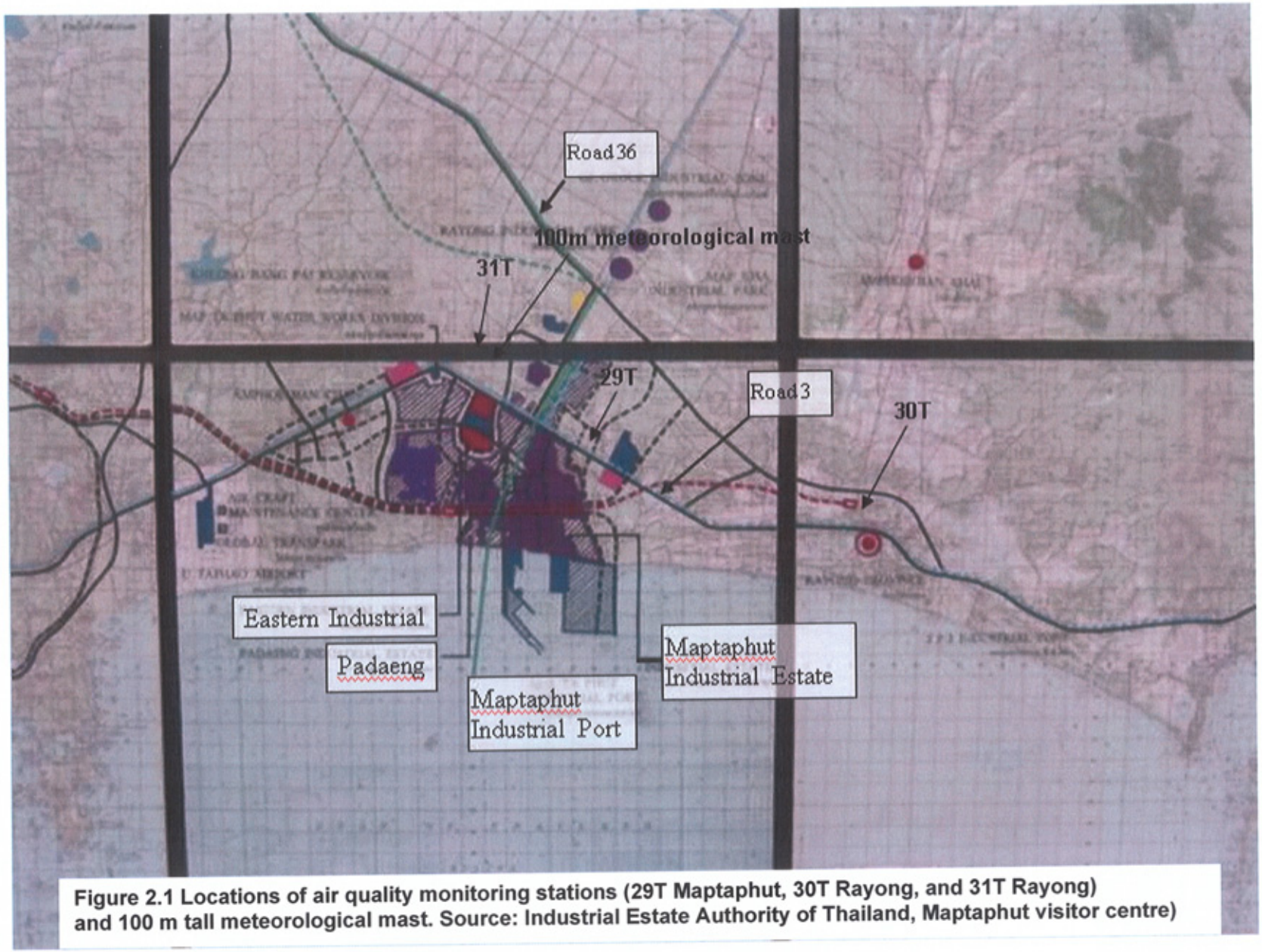

Figure 2.1 Locations of air quality monitoring stations (29T Maptaphut, 30T Rayong, and 31T Rayong) and 100 m tall meteorological mast. Source: Industrial Estate Authority of Thailand, Maptaphut visitor centre)

Table 2.3 Data collected from air quality monitoring stations and 100-metre meteorological mast

Station	Area type	Air pollutant (hourly basis)	Meteorological parameter* (hourly basis)
29T Maptaphut	Industrial/ Residential	CO, O_3, SO_2, PM_{10}, NO, NO_2, NOx, CH_4, NMHC	Surface: WS, WD, RH, T, P, NR, R
30T Rayong	Residential	CO, O_3, SO_2, PM_{10}, NO, NO_2, NOx, CH_4, NMHC	-
31T Rayong	Open area	CO, O_3, SO_2, PM_{10}, NO, NO_2, NOx, CH_4, NMHC	Surface: WS, WD, RH, T, P, GR, NR, R 30m: WD, RH, T
100m tall meteorological mast			Surface: WS, WD, RH, T, P, GR, NR, R 50m: WS, WD, T 100m: WS, WD, T

* WS-Wind Speed (m/s), WD-Wind Direction (Deg), RH-Relative Humidity (%), T-Temperature (°C), P-Pressure (mmHg), GR-Global Radiation (W m^{-2}), NR-Net Radiation (W m^{-2}), R-Rainfall (mm).

Figure 2.2 Ambient Air Monitoring Station 29T, Rayong Province.

Figure 2.3 Ambient Air Monitoring Station 30T, Rayong Province.

Figure 2.4 Ambient Air Monitoring Station 31T, Rayong Province.

Figure 2.5 Meteorological Instruments at 30-metre height of station 31T.

Figure 2.6 Meteorological Instruments at 10-metre height of station 31T.

Figure 2.7 100-metre Meteorological Tower behind the road (Route 3).

Figure 2.8 100-metre Meteorological Tower.

Figure 2.9 100-metre Meteorological Tower Base.

Figure 2.10 Temperature Sensor on 100-metre Meteorological Tower (2 m. height).

Figure 2.11 Global and Net Radiation Sensor Meteorological Tower (2, 10, 50, 100 m. height).

Figure 2.12 3D Ultra-Sonic Wind Sensor on 100-metre Meteorological Tower (10, 50, 100 m. height).

Table 2.4 Type of instruments used for pollutants monitoring

Parameters	Analytical Techniques	USEPA Reference (USEPA, 2008)
Carbon Monoxide (CO)	Gas Filter Correlation (GFC)	Gas Filter Correlation (GFC)
Ozone (O_3)	UV Absorption	UV Absorption
Sulphur Dioxide (SO_2)	UV Fluorescence	UV Fluorescence
Particulate Matter (PM-10)	β - Ray attenuation	β - Ray attenuation
Nitrogen Oxides (NO_x)	Chemiluminescense	Chemiluminescense
Hydrocarbon (CH_4, NMHC)	Flame Ionisation Detection	Flame Ionisation Detection

2.1.5 Instruments Used in Ambient Air Quality Monitoring Stations in Rayong Province

Instruments used in all ambient air quality monitoring stations normally consist of the analysers as listed above in Table 2.4. All of the above analytical techniques are recommended and approved by the US EPA. In addition to air quality analytical instruments, each station is equipped with a 10-metre meteorological tower, which has various sensors for ground meteorological conditions such as ambient air temperature, relative humidity, wind speed, wind direction, barometric pressure, rain, and net solar radiation. Figure 2.13 shows an example picture of the analysers. The techniques are reviewed in the following sections.

Figure 2.13 Data logger, NOx, CO, SO_2, HC analyser.

Gas Filter Correlation (GFC) CO Analyser (API, 2006a)

The detection and measurement of carbon monoxide is based on the absorption of infra red (IR) radiation by CO molecules at wave lengths near 4.7 microns. In practice, the analyser uses a high energy heated element to generate broad-band IR light. This light is passed through a rotating gas filter wheel, which causes the beam to alternately pass through a gas cell filled with nitrogen, (the measure cell) and a cell filled with CO/nitrogen mixture (the reference cell). This alternation occurs at a rate of 30 cycles per second and causes the beam to be modulated into reference and measure pulses. During a reference pulse, the CO in the gas filter wheel effectively strips the beam of all IR energy at wave lengths where CO can absorb. This results in a beam, that is unaffected by any CO in the sample cell. During the measure pulse, the nitrogen in the filter wheel does not affect the beam, which can subsequently be altered by any CO in the sample cell.

The gas filter wheel also incorporates an optical chopping mark, which superimposes a 360 cycles per second Light/Dark modulation on the IR beam. This high frequency modulation is included to maximise detector signal-to-noise performance.

After the gas filter wheel, the IR beam enters the multi-pass sample cell. This sample cell uses folding optics to generate a 16-metre absorption path length in order to achieve maximum sensitivity. Upon exiting the sample cell, the beam passes through a band-pass interference filter to limit the light to wave length of interest.

Finally, the beam strikes the detector, which is a thermoelectrically cooled solid-state photo-conductor. This detector, along with its pre-amplifier and bias voltage supply, converts the light signal into a modulated voltage signal. The detector output is electronically demodulated to generate two DC voltages, CO MEAS and CO REF. These voltages are proportional to the light intensity striking the detector during the measure pulse and reference pulse, respectively.

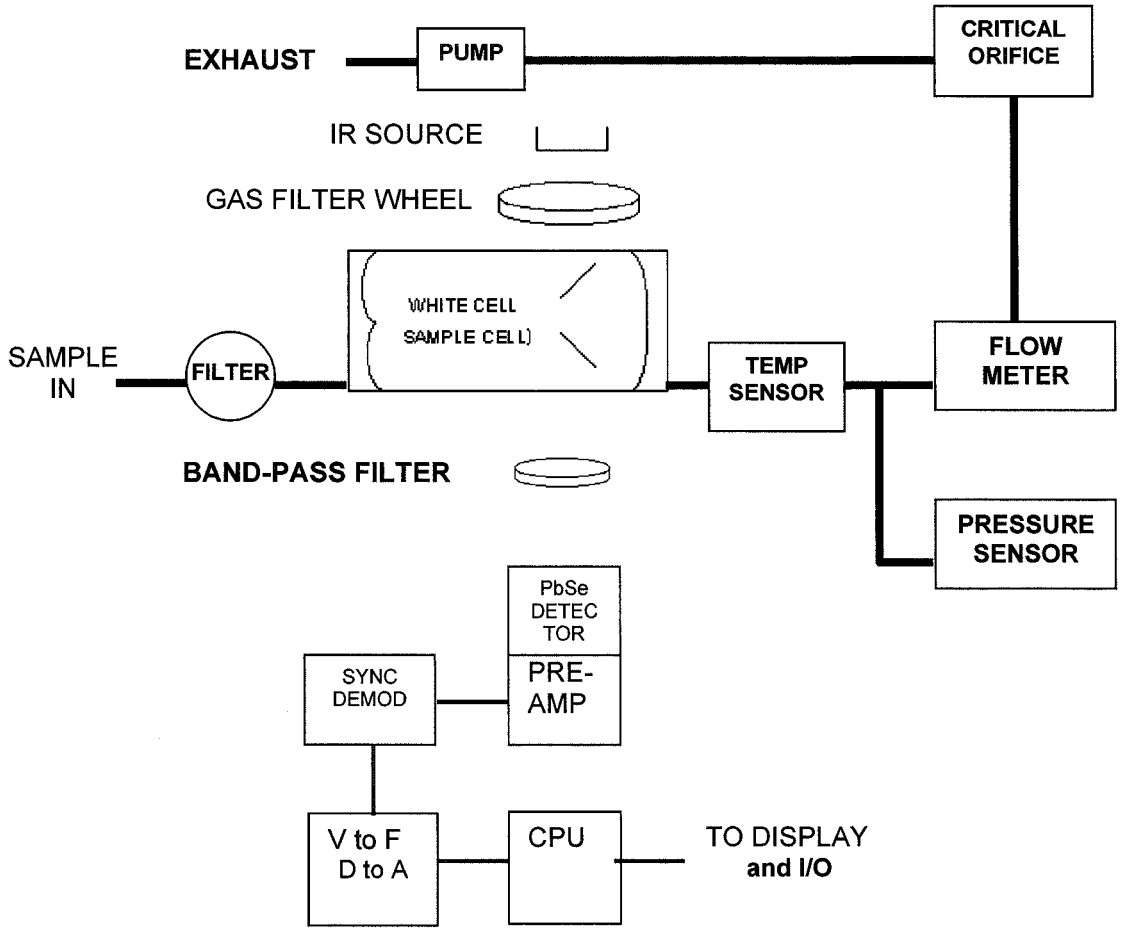

Figure 2.14 Carbon monoxide analyser.
Source: API Model 300, Gas Filter Correlation CO Analyser Operator Manual.

UV Absorption Ozone Analyser (API, 2007)

The detection of ozone molecules is based on absorption of 254 nm UV light due to an internal electronic resonance of the O_3 molecule. The analyser uses a mercury lamp constructed so that a large majority of the light emitted is at the 254 nm wavelength. Light from the lamp shines down a hollow glass tube that is alternately filled with sample gas, and then filled with gas scrubbed to remove ozone. The ratio of the intensity of light passing through the scrubbed gas to that of the sample forms a ratio I/I_o. This ratio forms the basis for the calculation of the ozone concentration.

The Beer-Lambert Equation, shown below, calculates the concentration of ozone from the ratio of light intensities.

$$c_{O_3}(ppm) = -\frac{1}{axL} x \frac{T}{273^\circ K} x \frac{29.92inHg}{P} x10^6 ppmxIn\frac{I}{I_\circ} \qquad \text{Equation 2.1}$$

Where:

I	=	Intensity of light passed through the sample
I_o	=	Intensity of light through sample free of ozone
A	=	absorption coefficient
L	=	path length
C	=	concentration of ozone in ppm
T	=	sample temperature in Kelvin
P	=	pressure in inches of mercury
In	=	natural logarithm

As can be seen, the concentration of ozone depends on more than the intensity ratio; temperature and pressure also influence the density of the sample. The density changes the number of ozone molecules in the absorption tube, which impacts the amount of light removed from the light beam. These effects are addressed by directly measuring temperature and pressure and including their actual values in the calculation. The absorption coefficient is a number that reflects the inherent ability of ozone to absorb 254 nm of light. Most current measurements place this value at 308 cm^{-1} atm^{-1} at STP. The value of this number reflects the fact that ozone is a very efficient absorber of UV radiation, which is why stratospheric ozone protects the life forms lower in the atmosphere from the harmful effects of solar UV radiation. Lastly, the absorption path-length determines how many molecules are present in the column of gas in the absorption tube. The intensity of

light is converted into a voltage by the detector/preamp module. The voltage is converted into a number by a voltage-to-frequency (V/F) converter capable of an 80,000 count resolution. The digitised intensities, along with the other variables, are used by the CPU to compute the concentration using the above formula.

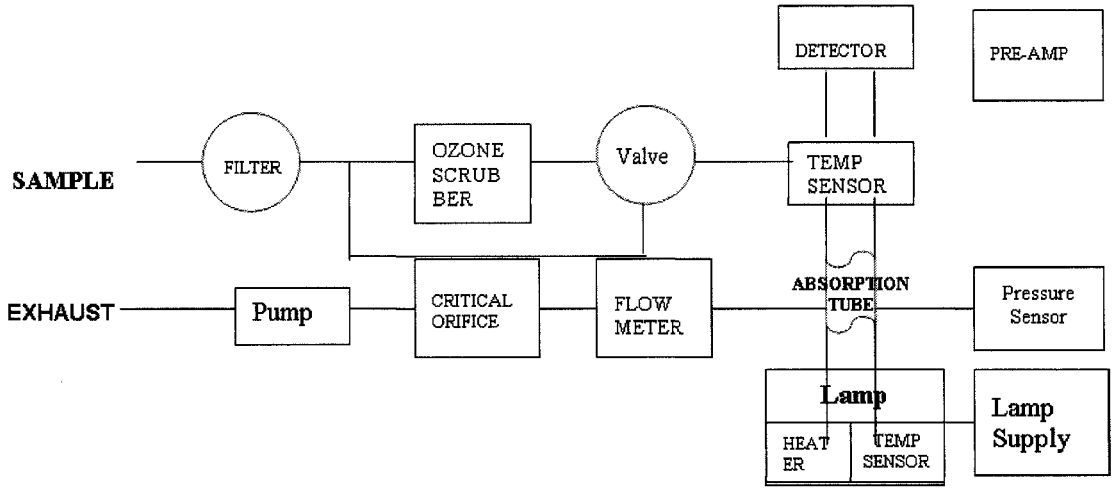

Figure 2.15 Ozone Analyser Block Diagram.
Source: API Model 400, Ozone Analyser Operator Manual.

UV Fluorescence SO_2 Analyser (API, 2006b)

The operation of the analyser is based upon the measurement of fluorescence of SO_2 due to absorption of UV energy. Sulphur dioxide absorbs in the 190nm-230nm region, free of quenching by air and relatively free of other interferences. Interferences caused by poly-nuclear aromatics are reduced by selective removal through a membrane without affecting the SO_2 concentration.

The UV lamp emits ultraviolet radiation, which passes through a 214nm filter (allowing 214nm light through), exciting the SO_2 molecules and producing fluorescence, which is measured by a photomultiplier tube with a 250 nm to 390nm filter. The Equations describing the above are as follows:

$$so_2 + h\nu_1 \rightarrow so_2^* \qquad \text{Equation 2.2}$$

The fraction of ultraviolet light absorbed, I_a / I_o, is given by:

$$I_a / I_\circ = e^{-abc} \qquad \text{Equation 2.3}$$

Where I_o is the initial UV light intensity, I_a is the intensity after absorption by SO_2, a is the absorption coefficient of SO_2, b is the path length, and c is the concentration of SO_2. The excited SO_2 decays back to the ground state emitting a characteristic fluorescence:

$$so_2^* \rightarrow so_2 + h\nu_2 \qquad \text{Equation 2.4}$$

When the SO_2 concentration is relatively low, the path length of excited light is short and the background is air, the fluorescent radiation impinging upon the PMT is directly proportional to the concentration of SO_2.

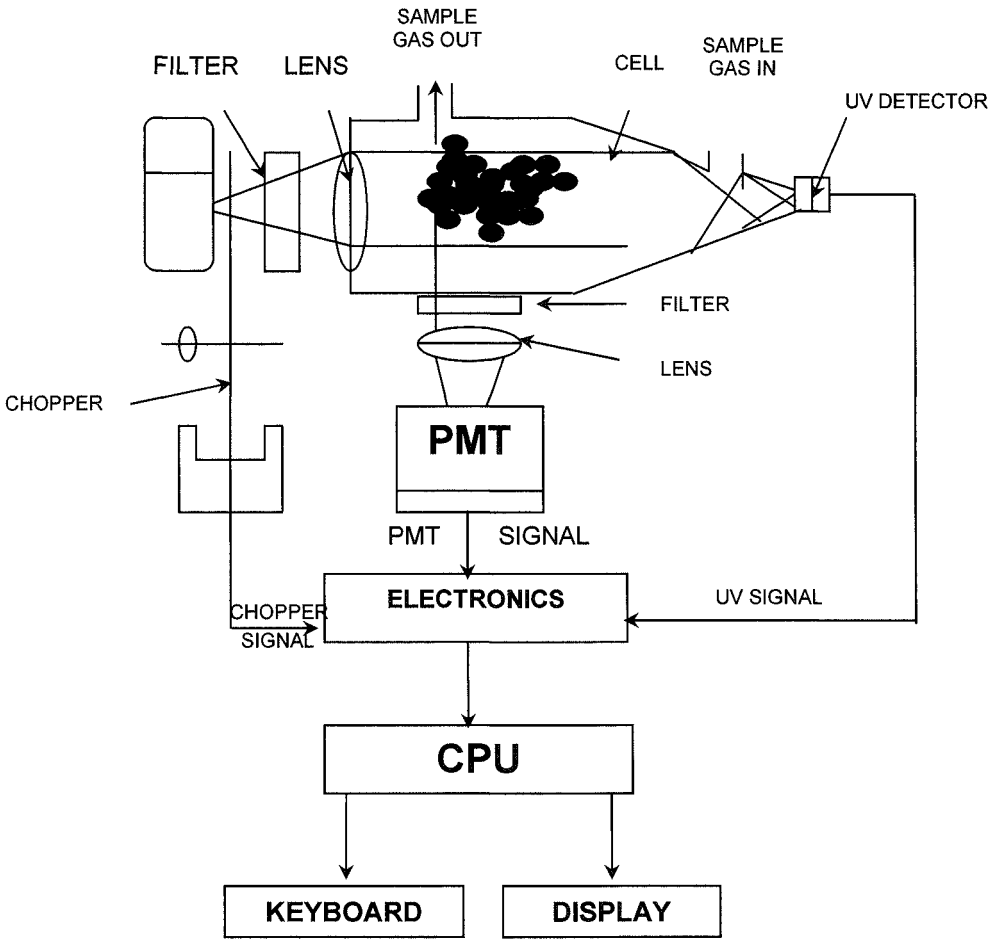

Figure 2.16 Principle of Operation.
Source: API Model 100, Fluorescent SO_2 Analyser Operator Manual.

β- Ray Attenuation PM-10 monitor (Met One Instruments, undated)
In the process of β decay, an electron is emitted from the atomic nucleus, and the nuclear charge changes from Z to Z + 1 in units of electronic charge. The electrons emitted by the β decay have energies ranging from zero to some well-defined maximum value E (M(Z,A)). This maximum energy is given by

$$E = [M(Z,A) - M(Z+1,A)]\,C^2 \qquad \text{Equation 2.5}$$

where M (Z,A) is the atomic mass of a nuclide with atomic number Z and atomic weight A, and c is the speed of light.

Beta particles interact with matter through elastic and inelastic scattering with atomic electrons and through elastic scattering with nuclei. At low energies, the absorption is mainly caused by inelastic ionising collisions with orbital electrons. When ionisation losses predominate, the attenuation of β particles becomes a function of the electron density of the absorbing medium. The electron density depends on the ratio of Z to A, which is nearly constant for all elements and most common isotopes, except for hydrogen. Most chemical compounds have a Z/A ratio in the range of 0.44 to 0.53. The penetration of low-energy β radiation depends almost exclusively on the area density (thickness times density) of the absorber and the maximum β energy of the impinging electrons, and is independent of the chemical composition of the absorbing matter.

The number of β particles passing through an absorber decreases exponentially with absorber thickness, to a good approximation, as given by the following:

$$I = I_o e^{-\mu X} \qquad \text{Equation 2.6}$$

Where I_o is the incident flux, I is the transmitted β flux through an absorber of area density x (mg/cm^2), and μ is the mass attenuation coefficient in cm^2 mg^{-1}. If μ is known from experimental calibration, then measurements of I and I_o can be directly related to the mass density of a given sample. The area density of particles collected on a filter tape at a constant flow rate, Q, for a sampling time, Δt, is given by

$$x(mg/cm^2) = \frac{Mc(\mu g/m^3) * Q(L/\min) * \Delta t(\min)}{10^6 * A(cm^2)} \qquad \text{Equation 2.7}$$

where Mc is the ambient mass concentration in μg m^{-3} and A is the particle deposit area in cm^2. Mc can be determined by

$$Mc = \frac{10^6 * A}{Q * \Delta t * \mu} \ell n(I_\circ / I) \qquad \text{Equation 2.8}$$

Equation 2.8 shows that accurate and stable measurement of both flow rate and β counts is essential for meaningful determination of ambient mass concentration.

The Mc values are determined using sampled air volumes at site atmospheric pressure and temperature. Should the user wish to express Mc in terms of reference condition of pressure (P_{ref}) and temperature (T_{ref}) as required by the U.S. EPA, the calculation must be performed manually, and the Equation for sampled air volume at reference conditions is given by :

$$Q_{ref} = Q * \frac{P_\circ}{T_\circ} * \frac{T_{ref}}{P_{ref}} \qquad \text{Equation 2.9}$$

Where Q is the flow rate at site atmospheric pressure, P_o, and temperature, T_o.

Chemiluminescence NO_x Analyser (API, 2005)

The Analyser is designed to measure the concentration of nitric oxide [NO], total oxides of nitrogen [NO_x] and, by calculation, nitrogen dioxide [NO_2]. The instrument measures the light intensity of the chemiluminescent gas phase reaction of nitric oxide [NO] and ozone [O_3] as follows:

$$NO + O_3 \rightarrow NO_2^* + O_2 \qquad \text{Equation 2.10}$$

$$NO_2^* \rightarrow NO_2 + h\nu \qquad \text{Equation 2.11}$$

The reaction between NO and ozone generates electronically excited NO_2 molecules as shown in the Equation 2.10 above. The excited NO_2 molecules release their excess energy by emitting a photon and dropping to a lower energy level as shown in the Equation 2.11. It has been shown that the light intensity produced is directly proportional to the [NO] concentration present.

The analyser samples the gas stream and measures the [NO] concentration by digitising the signal from the Analyser's photomultiplier tube (PMT). A valve then routes the sample

stream through a converter containing heated molybdenum to reduce any NO_2 present to NO by the following reaction:

$$3NO_2 + Mo \xrightarrow{315^\circ C} 3NO + MoO_3 \qquad \text{Equation 2.12}$$

The Analyser now measures the total NO_x concentration. The [NO_x] and [NO] values are subtracted from each other by the built-in computer yielding [NO_2]. The three results [NO], [NO_x], and [NO_2] are then further processed and stored by the computer yielding several instantaneous and long term averages of all three components.

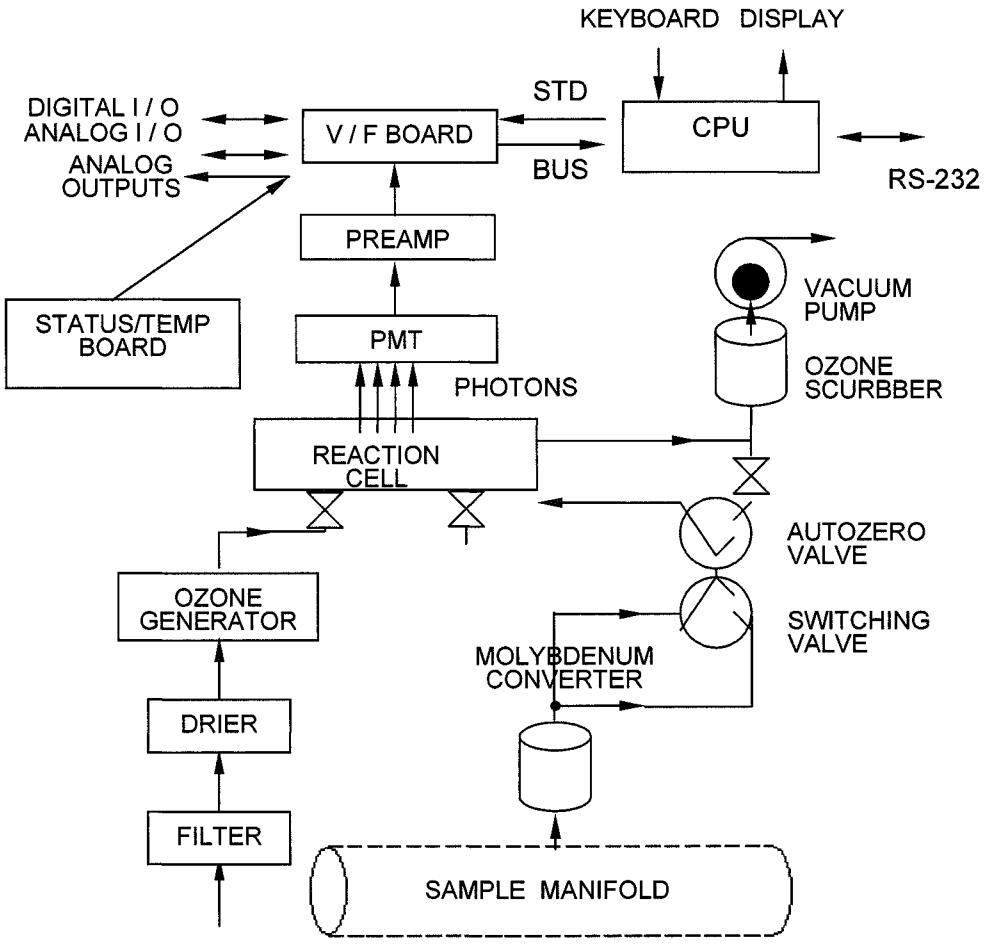

Figure 2.17 M 200A Nitrogen Oxides Analyser.

Source: API Model 200, Chemiluminescent NO/NO_2/NO_x Analyser Operator Manual.

Flame Ionisation Detection-Hydrocarbon Analyser (API, undated)

The Flame Ionisation Detector (FID) method is employed to determine the concentration of hydrocarbons present in a gaseous sample. Hydrogen burned in air will produce a flame containing a negligible number of ions. Introduction of a hydrocarbon sample into the flame results in a complex ionisation, creating a large number of ions. A polarizing voltage applied between the burner jet and a collector mounted above the flame, produces an electrostatic field that results in ion migration. Positive ions are attracted to the collector and negative ions to the burner jet, establishing an ionisation current between these two points. This current is directly proportional to the hydrocarbon concentration in the flame expressed as a carbon number. That is to say, that the output is proportional to the number of carbon molecules present in an organic form. The proportionality constant will vary with the type of C-H bonding, but for most purposes it is assumed that it is a single bond saturated.

A built in pump aspirates a sample into the instrument, where the flow is split and a small portion is routed to the burner at a constant flow rate, while the rest is bypassed through a precision backpressure controller to vent. At the burner, the sample is mixed with a controlled volume of hydrogen fuel and taken to the burner. A supply of hydrocarbon free air is also provided to the burner to support combustion. The flows and pressures of these gases is important since we are basically "counting carbon molecules" and anything which will affect the number present at the burner tip will affect the reading.

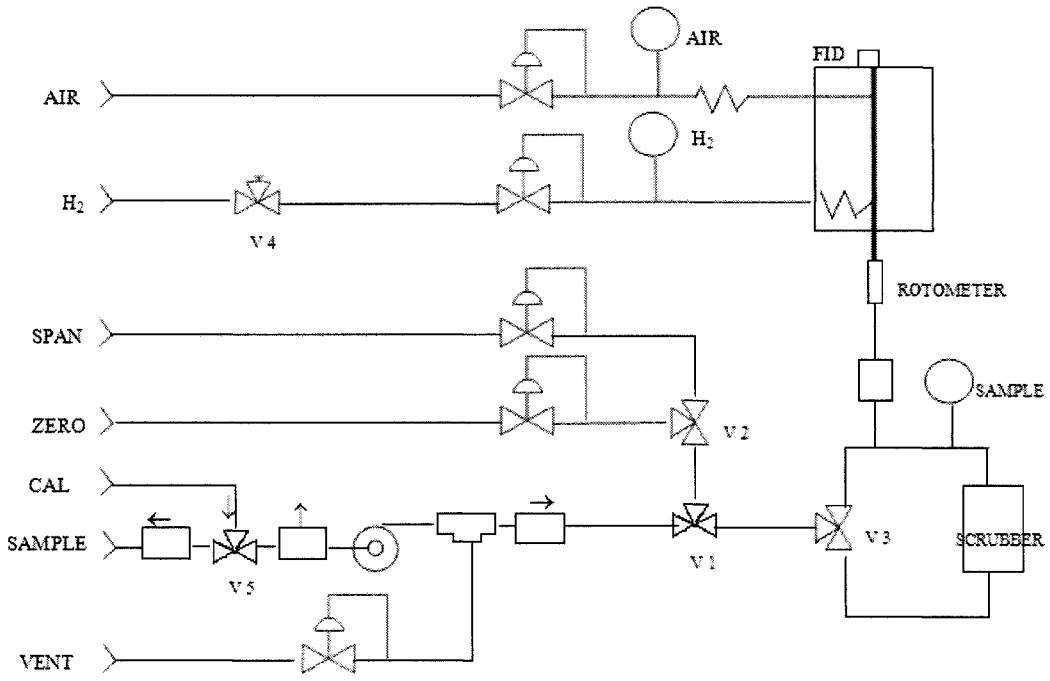

Figure 2.18 Hydrocarbon Analyser
Source: API manual

This will provide a reading of total hydrocarbons in ppm. Within the analyser, there is a series of flow switching valves controlled by a microprocessor. The sample flow is routed through a catalytic converter, which converts all hydrocarbons into carbon dioxide, hydrogen and associated components, but has no effect on the methane. The result of this converter, or scrubber, is to remove all hydrocarbons except the methane. The analyser is able to sequence the flow path to read total hydrocarbons, and total methane. It is then a simple measure to calculate the amount of total hydrocarbons, less methane.

Calibration of Analysers

The daily calibration of each analyser will be controlled through a data logger, which will command zero and span gas to be sent from zero air source and dilution calibrator, respectively. The dilution calibrator will dilute concentrated standard gases from the cylinder gas supply with zero air from zero air source to about 80% of the measuring span of each analyser. Single point calibration is done automatically on daily basis, where as multi point calibration is done every three months.

Pollutant gases are monitored continuously through the year. It is necessary to ensure that data captured is validated. Fig 2.19 and Fig 2.20 show how the ambient air quality monitoring station is connected between analysers, calibrator, zero air generator and data logger.

Daily calibration is performed automatically at 5.00 AM for approximately 30 minutes. During this process, zero and span calibration are carried out using zero gases from zero air generator and certified master gases from gas cylinder respectively. A calibrator is used to measure and control flow rate of zero air and span gases in order to dilute the certified gases to span level of each analyser. Once the zero gas and certified span gases are fed to each analyser, if the measurement is agreed within ± 5% then, all measurements for the rest of 24 hrs are considered to be valid (except power interruption or failure during 24 hrs period)

Multipoint calibration is performed every six months in order to ensure measurement linearity. In addition, single point and multipoint calibration are confirmed by external auditors using EPA Protocol gases every three months. It is also necessary to perform the following calibrations and checks to ensure good measurement: mass flow meter/controller of calibrator, meteorological sensors, conversion efficiency of Molybdenum converter of nitrogen oxides analyser, efficiency of oxidizer and non-methane cutter of hydrocarbon analyser. These are performed every six months.

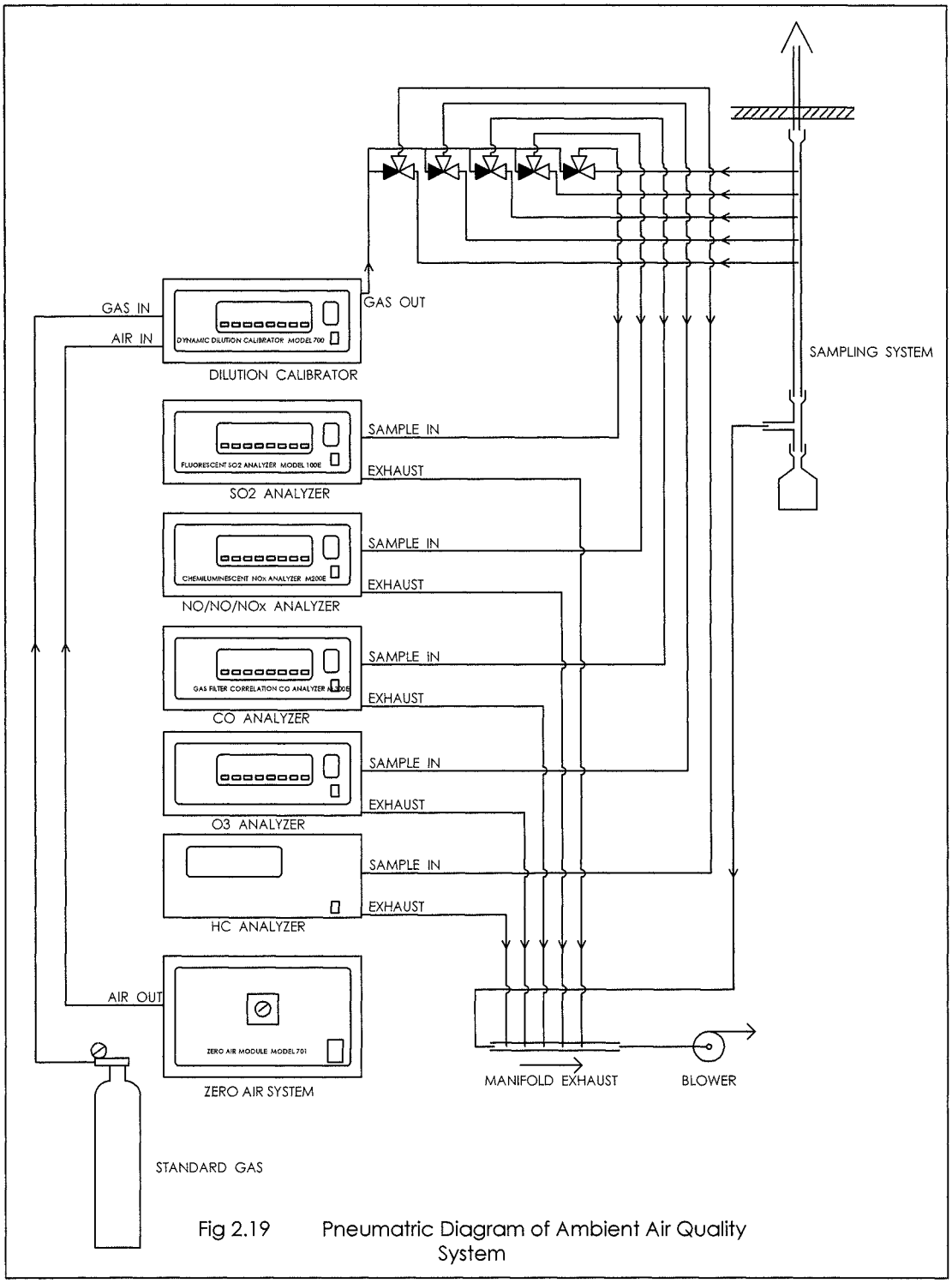

Fig 2.19 Pneumatric Diagram of Ambient Air Quality System

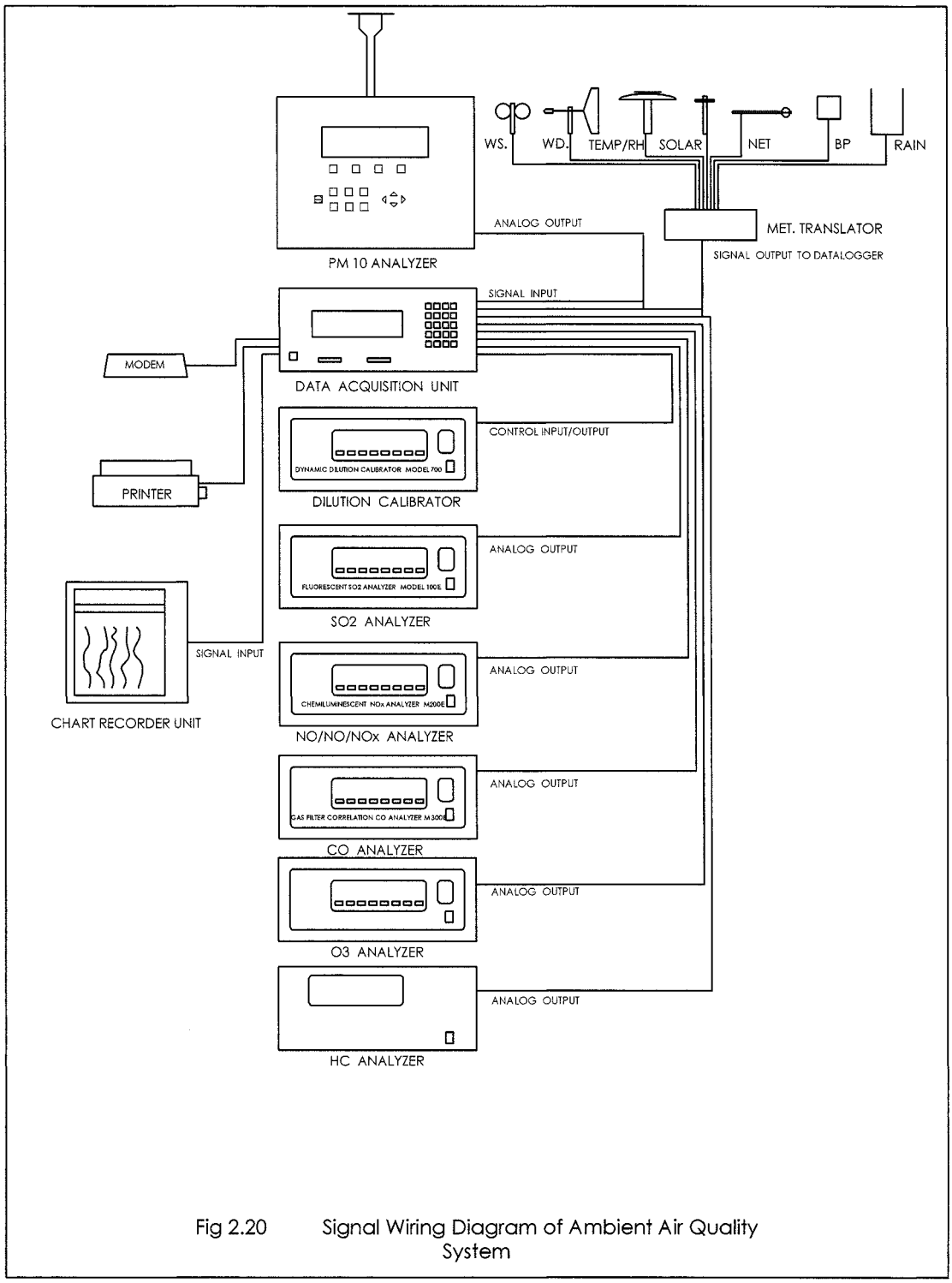

Fig 2.20 Signal Wiring Diagram of Ambient Air Quality System

Data logger

The data logger works as the brain of the ambient air quality monitoring stations. It has 16 channel analogue inputs with a memory of 258 kbytes. The retrieved data of each channel

is presented as an hourly average. Data status and faults will also be shown. In addition, the data logger controls on/off valves during the calibration process. Data stored in the data logger can be retrieved via modem on a PC located in the office. The format of raw ambient data and meteorological data recorded at PCD main data base computer is detailed in Appendix C.

2.2 Ambient Air Quality Standards

This research concerns the investigation of air quality and health impacts in Rayong Province, to the South and East of Bangkok; specifically the impact of the Maptaphut industrial estate on air quality and health in the local community will be investigated.

The first act for the control and conservation of national environmental quality in Thailand was the Enhancement and Conservation of National Environmental Quality Act of 1975, which established the Office of the National Environment Board (ONEB). The Act of 1975 and the ONEB were repealed and the Enhancement and Conservation of National Environmental Quality Act of 1992 was promulgated and published in the Royal Gazette on April 4, 1992. The Act became effective on June 4 of the same year. The Act of 1992 established three new offices under the Ministry of Science, Technology and Environment, namely the Office of Environmental Policy and Planning, The Pollution Control Department and the Department of Environmental Quality Promotion. At present, ambient air quality concern is the responsibility of the Ministry of National Resources and Environment, the Department of Pollution Control, and the Air Quality and Noise Management Division. The Air Quality and Noise Management Division is responsible for carrying out duties of the Department of Pollution Control with respect to air pollution, noise and vibration management. These duties include monitoring of air quality and noise, establishment of ambient air quality standards and emission standards, development of strategies to prevent and mitigate air and noise pollution, and preparation of reports (Wangwongwatana, 2001).

For the motor vehicle air pollution control, the relevant Acts are the Land Transport Act of 1992, The Motor Vehicle Act of 1979, the Traffic Act of 1992, the Announcement of the Revolutionary Party No. 16 of 1971, and the Liquid Fuel Act of 1978. The Land Transport Department, the Police Department, and the Department of Commercial Registration (The Office of the Council of State of Thailand, 1997) are responsible for implementing these Acts.

2.2.1. Ambient Air Quality Standards

In 1995, the 1981 National Primary Ambient Air Quality Standards (NPAAQS) were revised under S.32 of the Enhancement and Conservation of National Environmental Quality Act of 1992. The revised standards determined the latest information on human

health impact of key pollutants such as carbon monoxide (CO), nitrogen dioxide (NO_2), sulphur dioxide (SO_2), total suspended particulates (TSP), particulate matter (PM_{10}), lead (Pb), and ozone (O_3) within the constraints of specific environmental, socio-economic, and technological conditions that exist in Thailand. The reviewed standards are presented in Table 2.5 along with those for England and Wales, the USA and the World Health Organisation.

U.S. National Ambient Air Quality Standards (NAAQS) are standards established by the United States Environmental Protection Agency that apply for outdoor air throughout the country. Primary standards are designed to protect human health, including sensitive populations such as children, the elderly, and individuals suffering from respiratory disease. Secondary standards are designed to protect public welfare (e.g. building facades, visibility, crops, and domestic animals).

The present National Primary Ambient Air Quality Standards (NPAAQS) are more stringent than the 1981 standards, and are comparable to standards established in other nations and those recommended by relevant international organisations. The significant changes in Thai NPAAQS are the tighter values for the CO standards, both 1-hr and 8-hr averages, and the additions of a 1-hr average SO_2 standard, a 1-month average Pb standard, and 24-hr and annual average PM_{10} Standards for NO_2 and O_3 are left unchanged (Wangwongwatana, *et al.*, 2001)

Table 2.5 National Ambient Air Quality Standards

Pollutant	Averaging time (unit)	WHO Standards	Thai Standards	UK Objectives	US Standards
CO Carbon Monoxide	1-hour (ppm)	-	30.00	-	35.00
	1-hour (mg m^{-3})	30.00	34.20	-	40.00
	8-hour (ppm)	-	9.00	10.00	9.00
	8-hour (mg m^{-3})	10.00	10.26	11.60	10.00
NO_2 Nitrogen Dioxide	1-hour (ppm)	-	0.17	0.15	-
	1-hour (mg m^{-3})	0.20	0.32	0.287	-
	1-year (ppm)	-	-	-	0.053
	1-year (µg m^{-3})	40.00	-	-	100.00
O_3 Ozone	1-hour (ppm)	-	0.10	0.05	-
	1-hour (mg m^{-3})	-	0.20	0.10	-
	8-hour (ppm)	-	-	-	0.08
	8-hour (µg m^{-3})	120.00	-	-	
SO_2 Sulphur Dioxide	1-year (ppm)	-	0.04	-	0.03
	1-year (mg m^{-3})	0.05	0.10		
	24-hour (ppm)	-	0.12	-	0.14
	24-hour (mg m^{-3})	0.125	0.30	-	-
	1-hour (ppm)	-	0.30	-	-
	15-minute (ppb)	-	-	100.00	-
Pb Lead	1-month (µg m^{-3})	0.50	1.50	-	-
	Quarterly (µg m^{-3})	-	-	-	1.50
	1-Year (µg m^{-3})	-	-	0.25	-
PM-10 Particulate Matter (<10 µ)	24-hour (mg m^{-3})	-	0.12	0.05	0.15
	1-year (mg m^{-3})	-	0.05		0.05
Particulate Matter (< 100 µ)	24-hour (mg m^{-3})	-	0.33	-	-
	1-year (mg m^{-3})	-	0.10	-	-
$PM_{2.5}$ Fine Particulates	24-hour (µg m^{-3})	-	-	-	65.00
	1-year (µg m^{-3})	-	-	-	15.00

Source: Thailand (PCD, 2007), England (The Air Quality Strategy for England, Scotland, Wales and Northern Ireland. Department for the Environment Food and Rural Affairs in partnership with the Scottish Executive, The National Assembly for Wales and the Department of the Environmental for Northern Ireland, 2000), USA (USEPA, 2005)

2.3 Air Pollution Impacts on Public Health

This section reviews the health effects of ambient air pollutants.

2.3.1 Health and Welfare Effects of Exposure to NMHC and NO_x

Nitrogen oxides affect human health, particularly the respiratory system, by increasing the susceptibility to, and the severity of, lung infections and asthma in short-term and long-term exposure; these pollutants can also weaken the effectiveness of the lung's defences against bacterial infection (WHO, 1987). Nitrogen oxides are precursors for ozone formation and acidic precipitation, both of which harm terrestrial and aquatic ecosystems (Finlayson-Pitts *et al.*, 1986). Ozone presents a danger to human by irritating the mucous membranes of the nose, throat and airways in the respiratory system. When inhaled, 90% of the ozone remains in the lungs, only 10% is exhaled.

In smog, ground-level ozone is the main ingredient, and that is formed by a complex series of chemical reactions involving volatile organic compounds (VOC) and nitrogen oxides (NO_x) in the presence of heat and sunlight. Ozone is usually formed in the lower atmosphere during hot summer weather. There are a variety of sources of VOCs, including motor vehicles, chemical plants, refineries, factories consumer and commercial products, other industrial sources, and natural sources such as vegetation. Largely sources of NO_x are motor vehicles, non-road equipment, power plants, and other sources of combustion.

The chemistry of ozone formation, transport, and accumulation is complex. The ground-level ozone is generated and destroyed in a cyclical set of chemical reactions involving NO_x, VOC, heat and sunlight. In addition, CO also participates in the formation of ozone at much slower rate than most VOC and NO_x compounds, as shown in Equations 2.13 to 2.18.

$OH + CO \rightarrow H + CO_2$ Equation 2.13

$H + O_2 \rightarrow HO_2$ Equation 2.14

$HO_2 + NO \rightarrow OH + NO_2$ Equation 2.15

$NO_2 + hv \rightarrow NO + O$ Equation 2.16

$O + O_2 \rightarrow O_3$ Equation 2.17

The net effect of these reactions is:

$CO + 2O_2 \rightarrow CO_2 + O_3$ Equation 2.18

As a result of different emission of NO_x and VOC and patterns of weather, there are daily, seasonal, and yearly differences in ozone concentrations and differences from city to city. Many of the chemical reactions that are part of the ozone-forming cycle are sensitive to temperature and sunlight. When ambient temperatures and sunlight levels remain high for several days and the air is relatively stagnant, ozone and its precursors can build up and produce more ozone than typically would occur on a single high temperature day. In a further complication, ozone can be transported into an area from pollution sources found hundreds of miles upwind, resulting in elevated ozone levels even in areas with low VOC or NO_x emissions.

The EPA, based on a large number of recent studies, has identified several key health effects when people are exposed to the levels of ozone found today in many areas of the U.S. At high levels of ambient ozone concentrations, short-term exposures (1-3 hours) have been related to increased hospital admissions and emergency room visits for respiratory problems. For example, in the north-eastern U.S. and Canada the relationship between respiratory disease and the concentration of ozone was studied and indicated that the ozone air pollution was associated with 10-20 percent of all of the summertime respiratory-related hospital admissions. Exposure to ozone makes the respiratory infection and lung inflammation more serious; it can also aggravate pre-existing respiratory diseases, such as asthma. Moreover, exposure to ozone can cause repeated inflammation of the lung, impairment of the lung defence mechanisms, and irreversible changes in lung structure, which could lead to premature ageing of the lungs and/or chronic respiratory illnesses such as emphysema, chronic bronchitis and chronic asthma (EPA, 2003).

Children are most at risk, especially in summertime, because they are usually active outside. During the summer in the eastern U.S. and south-eastern Canada, ozone levels are the highest. It was reported that the children who were playing, exercising, and summer camping at that time show significant reductions in lung function. Children are also more at risk than adults from ozone exposure because their respiratory systems are still developing. During summer months, adults who are outdoor and moderately active, such as construction worker and outdoor worker, also are among those most at risk. The patients with respiratory illnesses such as asthma, particularly asthmatic children, can have impaired lung function and increased symptoms of respiratory illness, such as chest pains and coughs, when exposed to ozone during periods of moderate exertion (EPA, 2003).

There is evidence to indicate that daily increases in ozone levels and mortality levels are related. While the magnitude of this relationship was still too uncertain to allow for direct quantification, there are indications of likely positive relationships between ozone exposure and premature mortality (Jimmo, 2000).

Ozone not only affects human health, but is also known to adversely affect the environment in many ways, including: reduced yield for commodity crops, for fruits and vegetables, and commercial forests; ecosystem and vegetation effects in such areas as National Parks; damage to urban grass, flowers, shrubs, and trees; reduced yield in tree seeding and non-commercial forests; increased susceptibility of plants to pests; damage materials; and reduce visibility. A key precursor to ozone is nitrogen oxides (NO_x), which also results in nitrogen deposition into sensitive nitrogen-saturated coastal estuaries and ecosystems, causing increased growth of algae and other plants (Vitousek, 1997). Acid deposition is also contributed by NO_x; this phenomenon can damage trees at high elevations and increases the acidity of lakes and streams, which can severely damage aquatic life. Also, levels of particulate matter are increased by changing NO_x emissions into nitric acid in the atmosphere and forming particulate nitrate.

In addition to their contribution to ozone levels, emission of NMHC contains toxic air pollutants that also have a significant adverse effect on the public health.

2.3.2 Health and Welfare Effects of Exposure to Particulate Matter

Particulate matter (PM) comes from a variety of stationary and mobile anthropogenic sources. It represents a broad class of chemically and physically diverse substances that exist as discrete particles (liquid droplets or solids) over a wide range of sizes. Particle sources may be primary, directly emitted to the atmosphere, or secondary, formed by transformations of gaseous emissions such as sulphur dioxide or nitrogen oxides. The effects of PM are difficult to elucidate because the major chemical and physical properties of PM vary greatly with time, region, meteorology, and source category; thus, complicating the assessment of health and welfare effects as related to various indicators of particulate pollution. At high concentrations, particulate matter can adversely affect human health, visibility, and materials. Acid deposition is contributed by components of particulate matter e.g. sulphuric or nitric acid (U.S. EPA, 1996). Some of the main health effects are discussed as follows:

1) Particles can penetrate and deposit in the various regions of the respiratory tract, creating a biological response to these deposited materials.
2) During oronasal or mouth breathing, maximum ambient particles can deposit in the thorax (tracheobronchial and alveolar regions of the respiratory tract). The risks of adverse effects associated with deposition in the thorax are markedly greater than for deposition in the extrathoracic (head) region.
3) Most effects associated with ambient PM concentration, the key health effects, include premature death; aggravation of respiratory and cardiovascular disease, as indicated by increased hospital admissions and emergency room visits, school absences, work loss days, and restricted activity days; changes in lung function and increased respirator symptoms; changes to lung tissues and structure; and altered respiratory defence mechanisms. These key health effects have been used as a measure of population exposure, in a large number of community epidemiological studies. Additional information and insights on these effects are provided by studies of animal toxicology and controlled human exposures to various constituents of PM conducted at higher than ambient concentrations. Although mechanisms by which particles cause effects are not well known, there is general agreement that the cardio-respiratory system is the major target of PM effects.
4) Based on a qualitative assessment of the epidemiological evidence of effects associated with PM for populations that appear to be at greatest risk with respect to particular health endpoints, the EPA has concluded the following with respect to sensitive populations (US EPA, 1996):
    a) Individuals with respiratory disease (e.g., chronic obstructive pulmonary disease, acute bronchitis) and cardiovascular disease (e.g., ischemic heart disease) are at great risk of premature mortality and hospitalisation due to exposure to ambient PM.
    b) Individuals with Infectious respiratory disease (e.g., pneumonia) are at greater risk of premature mortality and morbidity (e.g., hospitalisation, aggravation of respiratory symptoms) due to exposure to ambient PM. Also, exposure to PM may increase individuals' susceptibility to respiratory infections.
    c) Elderly individuals are also at greater risk of premature mortality and hospitalisation for cardiopulmonary problems due to ambient PM
    d) Children are at greater risk of increased respiratory symptoms and decreased lung function due to exposure to ambient PM.

e) Asthmatic individuals are at risk of exacerbation of symptoms associated with asthma, and increased need for medical attention, due to exposure to PM.

5) Fine and coarse fraction particles are different in fundamental physical and chemical properties. The difference relates to the nature of potential effects and relative concentrations required to produce such effects. The fine fraction is the specific components of PM that could be concern to health such as acid aerosols, sulphates, nitrates, transition metals, diesel particles, and ultra fine particles. Examples of the coarse fraction of PM are silica, resuspended dust, and others. Both fractions can cause health effects but the epidemiological studies have indicated that the fine fraction appears to contain more of the reactive substances capable of causing adverse effects. In addition, the fine fraction contains the largest number of particles and a much larger aggregate surface area than the coarse fraction, which enables a substantially greater potential for absorption and deposition in the thoracic region as well as for dissolution or absorption of pollutant gases.

There are also secondary effects of particulate matter, such as: impairment of visibility over urban areas and large multi-state regions; materials damage;, and soiling and acid deposition.

In all regions of the United States, particulate pollution is a problem affecting localities in both urban and non-urban. Airborne particulate matter that results from manmade emissions principally are from stationary point sources (fuel combustion and industrial processes), industrial process fugitive particulate emission sources, non-industrial fugitive sources (roadway dust from paved and unpaved roads, wind erosion from crop land, etc.), and transportation sources. Moreover, natural emissions also generate particulates including dust, sea spray, volcanic emissions, biogenic emanation (e.g., pollen from plants), and emissions from wild fires.

2.3.3 Health and Welfare Effects of Exposure to Carbon monoxide

When inhaled, carbon monoxide (CO) accumulates in the blood and forms a toxic compound known as carboxyhaemoglobin that prohibits oxygen transport in the human organism. Carboxyhaemoglobin can cause headaches, fatigue, nausea, dizzy spells,

confusion and irritability or flu like symptoms. Later stages of CO poisoning can cause vomiting, loss of consciousness and eventually brain damage or death (WHO, 2000).

2.3.4 Review of Health Impact Data.

Many researchers have used statistical approaches to verify the health impact of air pollution. The general conclusions from these studies are:

1) The levels of ambient air pollution make a significant contribution to the variation in daily hospital admissions for asthma and respiratory disease.
2) The adverse effect of air pollution on health exists even when levels of pollutants are still lower than standard values.
3) The air pollution has associations with daily respiratory admissions, cardio respiratory admissions to hospital and cerebrovascular admissions, admissions for chronic obstructive pulmonary disease (COPD) and bronchitis admissions.

Overall, the evidence points to a causal relationship, but the mechanism of action, exposure response relationships and pollutant interactions remain unclear. Definitive conclusions can be justified only after meta-analysis. The causal interpretation of reported associations between daily air pollution and daily admissions requires consideration of residual confounding, correlation between pollutants, and effect modification. One of the interesting results suggested is that aeroallergens might be statistically associated more strongly with asthma hospital admissions than air pollutants and may act as confounding factors in epidemiological studies (Rosas *et al*, 1998).

In this section epidemiological studies are grouped together in terms of the epidemiological approach adopted, and for each approach the studies are discussed in chronological order. Many of the studies have analysed the statistical significance of health effects as a result of exposure to several air pollutants at the same time, i.e. ambient exposure, and so the studies are not further subdivided along pollutant lines.

Epidemiological investigations can be categorised firstly into short term (acute) and long term (chronic) studies; short term methods include time series analysis, panel studies and meta-analysis; long term methods can be divided principally into cohort studies and intervention studies (WHO, 2005).

2.3.4.1 Time series analysis.

Time series analyses probably represent the most abundant of the health impact studies that have been carried out, and have contributed greatly to the understanding of short-term health impacts of air pollutants. Essentially these studies analyse the statistical variation observed between specific measures of ill health (end points) and exposure to measured concentrations of ambient air pollutants; meteorological conditions are also measured (WHO, 2005). Usually the studies will be city or location specific, though meta analysis can be used to analyse pooled data from several independent studies conducted in similar geographical locations (see section 2.3.4.2). Statistical analysis, such as multiple linear regression (MLR) can identify significant pollutants and climatic variables from among the overall data set (the work described later in this thesis will seek to carry out a time series analysis based on weekly hospital admissions for respiratory illness in the Maptaphut and Rayong areas - Section 3.7).

Evidence of the effect of carbon monoxide health impacts comes from a study carried out by Kurt *et al* (1978). Here it was suggested that the frequency of thoracic complaints in an emergency room can be affected by the ambient level of CO.

A non-association was found in a study carried out in Southern Ontario, where a 6-year data set of daily counts of admissions to 79 acute care hospitals was analysed in relation to concurrent measurements of air pollution and weather pooled over the same regions (Lipfert and Hammerstrom, 1992). Twenty-four-hour averages for air quality were found to yield more significant associations than peak hourly concentrations. Using cumulative lags increased the magnitude of the estimated response to about 20% of summer respiratory admissions, but no consistent relationships was found which could identify the "responsible" pollutant(s) with certainty. Average pollutant concentrations were generally within U.S. ambient standards.

In a longer term study carried out in Hong Kong, Tseng *et al* (1992) examined the relationship between concentrations of SO_2, O_3, Total Suspended Particulates (TSP), Respirable Suspended Particulates (RSP), NO_2, and NO_X and quarterly asthmatic hospital discharge rates during 1983 to 198. Discharges from the hospital of children under 14 years of age represented 56% of the 33,952 discharges recorded for all age groups. Trends of adult hospitalisation rates over time remained stable during the study period. In children, however, there was an increase in these rates, particularly marked in the age

group of one to four years. Univariate analysis revealed a strong correlation between quarterly mean TSP and hospital discharge rate for the one to four-year-old children (r = 0.62, P less than 0.001). In the five to 14-year-old age group, there was an inverse relationship between hospital discharge rate and sulphur dioxide level (r = -.38, P less than .05). Stepwise multiple regression analysis, with controlling for confounding variables (seasonal and annual trends of asthma hospitalisations) confirmed these relationships. A highly significant linear regression equation was derived between hospitalisation rate for ages one to four years and total suspended particles (P less than .001). The highly significant correlation between pollution and asthmatic hospitalisation rate for the one to four-year-old group suggested that young children are vulnerable to the adverse environmental effects of pollution.

A retrospective study using ambient O_3, temperature, and other environmental variables and their effects on the frequency of hospital visits for asthma was conducted in New Jersey, an area that often exceeded the allowable national standard for O_3 (Cody, et. al., 1992). Their finding showed the contribution of ozone to asthma admissions to be stronger in areas with higher ozone concentrations.

Schwartz *et al* (1993) also looked exacerbation of asthma, by elevated pollutant levels. The study of daily records of asthma emergency room visits from eight hospitals in Seattle, US, were used. In Poisson regressions, controlling for weather, season, time trends, age, hospital, and day of the week, the daily counts of emergency room visits for persons under age 65 were significantly associated with PM_{10} exposure on the previous day. The mean of the previous four days' PM_{10} was a better predictor ($p < 0.005$). The relative risk for a 30 $\mu g\ m^{-3}$ increase in PM_{10} was 1.12 (95% confidence interval 1.20 to 1.04). Daily PM_{10} concentrations never exceeded 70% of the current ambient air quality standards during the period.

Another study of the effect of pollutants on asthma admissions was carried out by Rossi, *et al* (1993), this time indicating NO_2 as a risk factor in addition to, but to a greater extent than particulate matter. The number of daily attendances for asthma attacks at the emergency room at Oulu University Central Hospital (Finland), was recorded over one year together with daily meteorological readings (temperature, humidity, barometric pressure, rainfall), levels of air pollutants (NO_2, SO_2, H_2S, TSP), and pollen counts (birch, alder, pine, willow, total pollen). Multiple regression and stepwise discriminate analysis were used to analyse the relationship between the number of attendances and the

measured variables. The total number of attendances during the year was 232, with lower figures in summer and higher in winter. No association was found between visits for asthma attacks and airborne pollen levels or meteorological factors except for temperature, which had a low inverse correlation with attendance. The most significant correlations were found between asthma visits and the levels of NO_2; those for SO_2, TSP, and H_2S were also significant. Intercorrelations between SO_2 and temperature or NO_2 and between temperature and TSP or NO_2 were also found, but only NO_2 correlated significantly with attendances after standardisation of temperature. Increased levels of pollutants, especially NO_2, were associated with attacks of asthma, but the explanation for this is unobvious. Air pollen levels were not associated with asthma attacks and only temperature among the meteorological factors had a small association with asthma.

A UK-based study carried out by Walters *et al* (1994) looked at whether daily and weekly variations in the levels of smoke and SO_2 in Birmingham are related to hospital admissions for asthma and acute respiratory diseases; both pollutants were indicated to present significant health risks. Daily numbers of hospital admissions for asthma and acute respiratory conditions for residents of Birmingham between 1988 and 1990 were obtained from West Midlands RHA Korner inpatient data. Average daily levels of SO_2 and smoke were taken from Birmingham City Council for the same period, together with daily meteorological summaries from the Department of Geography, University of Birmingham. Data was divided into seasons and the relation between hospital admissions and pollutant levels analysed by stepwise least squares regression models. Meteorological variables (temperature, pressure, humidity) were entered into the model if they showed significant association with hospital admissions during the season in question. Analysis was undertaken for daily (same day and lagged by two days) and weekly pollutant levels. Admissions were lagged behind pollution levels to allow for delayed effects of pollutants. The mean daily level of smoke was 12.7 $\mu g\ m^{-3}$ and of SO_2 was 39.1 $\mu g\ m^{-3}$, with maxima of 188.3 $\mu g\ m^{-3}$ and 126.3 $\mu g\ m^{-3}$, respectively. Significant associations were found between hospital admissions for respiratory disease lagged by two days, and smoke and SO_2 levels during winter. Associations between admissions for asthma and smoke and SO_2 levels were significant at the 5% level. These were independent of temperature, pressure, and humidity. Stepwise regression including both pollutants showed that smoke, but not SO_2, was a significant independent predictor of hospital admissions for both asthma and all respiratory conditions. During winter a rise of 100 $\mu g\ m^{-3}$ smoke might result in five (95% CI 0.6 to 9) more asthma admissions and 21.5 (95% CI 10 to 33) more

acute respiratory admissions each day in Birmingham. A 100 $\mu g\ m^{-3}$ rise in SO_2 might result in four (0 to 7) more asthma admissions and 15.5 (6 to 25) more respiratory admissions each day. Independent associations were also found between weekly mean smoke and SO_2 levels and all respiratory admissions during autumn and winter. During summer, daily mean smoke and SO_2 levels were significantly associated with non-lagged daily admissions for all respiratory diseases ($p < 0.02$). During spring, there was no association between air pollution and hospital admissions. Daily variations in smoke and SO_2 levels are significantly associated with hospital admissions for asthma and respiratory disease during winter in Birmingham. This association was independent of potential confounding effects of weather (temperature, pressure, humidity) and suggests that current levels of air pollution can still produce significant health effects.

Short-term effects of air pollution on morbidity were studied for data collected for Athens, (Pantazopoulou, *et al,* 1995). Data was collected on the daily number of emergency outpatient visits and admissions for cardiac and respiratory causes to all major hospitals in the greater Athens area during 1988. Measurements of air pollution included values for smoke, CO and NO_2. Statistical analysis was carried out using multiple linear regression models adjusted for potential confounding effects of meteorological and chronological variables, separately for winter (1/1-3/21 and 9/22-12/31) and summer (3/22-9/21). The number of emergency admissions for cardiac and respiratory causes was related to all indices of air pollution during the winter. Thus, the average adjusted increase in the daily number of cardiac admissions corresponding to an increase from the 5th to the 95th percentile of the season-specific distribution of each pollutant ranged from 15 to 17% or from 11 to 12.5 admissions per day, and for the daily number of respiratory admissions from 20 to 29% or from 8.3 to 12.1 admissions. The results of the present study indicate that air pollution in the Athens area has short-term effects on morbidity in the population.

A time series analysis carried out in Finland in the late 1980's showed that symptoms of ischemic cardiac and cerebrovascular diseases may be provoked by pollutants in concentrations lower than those given as guidelines in many countries. The study was carried out to determine whether daily concentrations of SO_2, NO, NO_2, O_3, and particulates are related to daily hospital admissions due to ischemic cardiac and cerebrovascular diseases in Helsinki (Ponka and Virtanen, 1996a). Admissions via emergency rooms due to ischemic cardiac diseases ($n = 7005$) were significantly associated with the prevailing levels of NO and O_3, and those due to cerebrovascular diseases ($n = 3,737$) were associated with NO_2. These levels were only moderate. Long-

term transient myocardial ischemic attacks were related to particulates, and short-term ischemic attacks were related to NO_2.

The same authors analysed the relationships between the daily number of asthma admissions and air pollutants were analysed with Poisson regression, taking into account potential confounding factors by using standardised protocol of the APHEA project at Helsinki, Finland, 1987-1989 (Ponka and Virtanen, 1996b). Positive relationships with admissions were observed for O_3 levels in all children under 14 years, and for SO_2 levels in 15-64 years old and among those older than 64. Significant associations were also seen between admissions for digestive tract diseases (the control) and O_3 levels. This suggests that the modelling, which proved to be problematic, was unsatisfactory, or it may be a statistical coincidence. A rise in temperature was associated with a low number of admissions for asthma among 0-14 year olds and among 15-64 year olds, whereas humidity did not have a significant effect on the number of admissions.

A study analysed the relationship between pollutant concentrations and the daily hospital admissions in 1988 at Latrobe Valley, Victoria (Voigt, Bailey and Abramson, 1998). There were associations ($r = 0.11$ to 0.17) between airborne particles, NO_2 and respiratory morbidity. There was no significant relationship between any of the pollutants and asthma admissions. However, multi-variate analysis confirmed that NO_2 and particulates were associated with admissions for COAD (Chronic Obstructive Airways Disease).

In Mexico City, a statistical analysis report was compiled on the relationship between emergency admissions for asthma to a hospital and daily average airborne concentrations of pollen, fungal spores, air pollutants (O_3, NO_2, SO_2, and particulates) and weather factors (Rosas, et. al., 1998). Asthma admissions had a seasonal pattern with more during the wet season (May-October) than the dry season (November-April). There were few statistical associations between asthma admissions and air pollutants for the three age groups studied in either season (children under 15 years, adults, and seniors [adults over 59 years]). Grass pollen was associated with child and adult admissions for both the wet and dry seasons, and fungal spores were associated with child admissions during both the wet and dry seasons. The analysis was done with environmental data averaged over the day of admission and the two previous days. Their results suggested that aeroallergens may be statistically associated more strongly with asthma hospital admissions than air pollutants and may act as confounding factors in epidemiological studies.

In Singapore, data for ambient air-pollutant levels, meteorological factors, and hospitalisation or emergency room visits for acute asthma children over a five-year period (1990-1994) were obtained and analysed for associations by time-series methods (Chew, *et al* 1999). Associations were observed even though the overall levels of all pollutants were generally within the air-quality guidelines established by the WHO. These findings suggest that asthmatic children are susceptible to increase levels of air pollutants, particularly SO_2 and TSP, even though the ambient levels are generally within acceptable ranges.

In another study carried out in South East Asia, this time in Bangkok, a tropical city of over six million people over PM10 was found to be significantly associated with increased mortality rates. The study was based on the comparison of mortality rates with hourly monitoring data fir ambient PM_{10}. In this metropolitan area, PM_{10} consisted largely of fine particles generated from diesel- and gasoline-powered automobiles, and from two-stroke motorcycle engines. The analysis involved the examination of the relationship between PM_{10} and daily mortality for 1992 through 1995 (Ostro, *et. al.,* 1999). In addition to counts of daily natural mortality (total mortality net of accidents, homicides, and suicides), the data was compiled to assess both cardiovascular and respiratory mortality, and natural mortality by age group. A multivariate Poisson regression model was used to explain daily mortality while controlling several covariates including temperature, humidity, day of the week, season, and time. The analysis indicated a statistically significant association between PM_{10} and all of the alternative measures of mortality. The results suggested a 10 $\mu g\ m^{-3}$ change in daily PM_{10} was associated with a 1-2% increase in natural mortality, a 1-2% increase in cardiovascular mortality, and a 3-6% increase in respiratory mortality.

In Santiago, Chile, a time-series analyses study was conducted for two years of daily visits to primary health care clinics. Counts were computed for either upper or lower respiratory symptoms and for cohorts of children 3-15 years of age and below age two (Ostro, *et.al.,* 1999). Daily PM_{10} and O_3 measurements and meteorological variables were available from instruments located in Santiago. The multiple regression analysis indicated a statistically significant association between PM_{10} and medical visits for lower respiratory symptoms in children ages 3-15 and in children under age two. PM_{10} was also associated with medical visits related to upper respiratory symptoms in the older cohort, while ozone was associated with visits related to both lower and upper respiratory symptoms in the older cohort. For children under age two, a 50 $\mu g\ m^{-3}$ change in PM_{10} (the approximate

interquartile range) was associated with a four-12% increase in lower respiratory symptoms. For children 3-15 years of age, the increase in lower respiratory symptoms ranges from 3 to 9% for a 50 $\mu g\ m^{-3}$ change in PM_{10} and 5% per 50 ppb change in O_3

In an elaborate study from North America, a wide variety of respiratory and cardiac conditions and an extensive set of pollutants and aeroallergens were considered, and utilised prospectively collected information on possible effect modifiers (Stieb, *et. al.*, 2000). At Saint John, the relationship between air pollution, aeroallergens and cardiorespiratory ED (Emergency Department) visits (n = 19,821) was examined for the period 1992 to 1996 using generalised additive models. ED visits, air pollution and aeroallergen time series were prefiltered using LOESS (Locally Weighted Regression) smoothers to minimize temporal error, and a developed model was constructed to reduce error from weather and days of the week. Multi-pollutant and multi-aeroallergen models were constructed using stepwise procedures and sensitivity analyses were conducted by season, diagnosis, and selected individual characteristics or effect modifiers. In single-pollutant models, positive effects of all pollutants except NO_2 and CO were observed on asthma visits, and positive effects on all respiratory diagnosis groups were observed for O_3, SO_2, PM_{10}, $PM_{2.5}$, and SO_2. Among cardiac conditions, only dysrhythmia visits were positively associated with all measures of particulate matter. In the final year-round multi-pollutant models, a 20.9% increase in cardiac ED visits was attributed to the combination of O_3 (16.0%, 95% CI 2.8-30.9) and SO_2 (4.9%, 95%CI 1.7-8.2) at the mean concentration of each pollutant. In the final multipollutant model for respiratory visits, O_3 accounted for 3.9% of visits (95% CI 0.8-7.2), and SO_2 for 3.7% (95% CI 1.5-6.0), whereas a weak, negative association was observed with NO_2. In multi-aeroallergen models of warm season asthma ED visits, Ascomycetes, Alternaria and small round fungal spores accounted for 4.5% (95% CI 1.8-7.4), 4.7% (95% CI 1.0-8.6) and 3.0% (95% CI 0.8-5.1), respectively, of visits at their mean concentrations, and these effects were not sensitive to adjustment for air pollution effects. Although in single-pollutant models, positive associations were noted between ED visits and some measures of particulate matter, in multi-pollutant models, pollutant gases, particularly ozone, exhibited more consistent effects. Aeroallergens were also significantly associated with warm season asthma ED visits.

Not all time series analyses have shown statistically significant associations. For example: the associate effects of fine particles ($PM_{2.5}$), coarse particles (PM2.5-10), black smoke (mainly fine particles of primary origin) and sulphate (mainly fine particles of secondary

origin) were investigated, together with ozone, SO_2, NO_2, and CO, on daily mortality and hospital admissions in the West Midlands conurbation of the United Kingdom (Anderson, *et.al.*, 2001). Clear effects of air pollution on mortality and hospital admissions were difficult to elucidate, except in certain age or diagnostic subgroups and seasonal analyses. It was also difficult to distinguish between different measures of particles.

The effects of ambient air pollution on hospital admissions in Brisbane, Australia were investigated: the European approach protocol was used to examine the effects of particles, O_3, SO_2, and NO_2 on daily hospital admissions for asthma and respiratory, cardiovascular, and digestive disorders (control diagnosis) that occurred during the period 1987-1994 (Petroeschevsky, *et. al.*, 2001). O_3 was consistently associated with admissions for asthma and respiratory disease with little evidence of a threshold. In two-pollutant models, the O_3 effect was unaffected by reducing other pollutants. Particulate pollution (measured by nephelometry) was associated positively with admissions for respiratory disease and admissions for asthma in summer, whereas a negative association was observed for cardiovascular admissions. Although SO_2 was associated significantly with admissions for respiratory and cardiovascular diseases, a significant association was also found for the control diagnosis of digestive disorders. No significant association was found for NO_2 over the study period, although significantly positive seasonal interactions were found for asthma and respiratory disease in autumn, winter, and spring. It was concluded that current levels of ambient air pollution in Brisbane make a significant contribution to the variation in daily hospital admissions for asthma and respiratory disease.

A study carried out in London examined the relationship between air pollution and upper respiratory disease for families attending doctor's surgeries (Hajat, *et. al.*, 2002). The study used non-parametric methods of analysis of time series data, adjusting for seasonal factors, day of the week, holiday effects, influenza, weather, pollen concentrations, and serial correlation. It was estimated that a 10-90th percentile change (13-31 $\mu g\ m^{-3}$) in SO_2 concentration resulted in a small increase in the number of childhood consultations: 3.5% (95% CI 1.4% to 5.8%). Stronger associations were found in the case of a 10-90th percentile change (16-47 $\mu g\ m^{-3}$) in fine particles PM_{10} in adults aged 15-64: 5.7% (95% CI 2.9% to 8.6%), and in adults aged 65 and over: 10.2% (95% CI 5.3% to 15.3%). Generally, correlations were strongest for elderly people, weakest in children, and were largely found in the winter months for these two age groups, and in the summer months for adults aged

15-64. The results provide strong evidence for an adverse effect of air pollution on consultations for upper respiratory symptoms, in particular in the case of PM_{10} and SO_2.

A study into health effects of PM_{10}, NO_2, SO_2, and O_3 for two cities of very different climates, lifestyles, and many other respects (Wong et. al., 2002). The selected cities were Hong Kong and London. In each city, associations between daily admissions and pollutant levels were analysed by the Poisson regression statistical methods. In addition, non-parametric smoothing methods were used to model seasonality and non-linear dependence of admissions on temperature, humidity, and influenza admissions. For respiratory admissions (> or = 65 years of age), significant positive associations were observed with PM_{10}, NO_2, SO_2, and O_3 in both cities. These associations tended to be stronger at shorter lag times in Hong Kong and at longer lag times in London. Associations were stronger in the cool season in Hong Kong and in the warm season in London, periods during which levels of humidity are at their lowest in each city. For cardiac admissions (all ages) in both cities, significant positive associations were observed for PM_{10}, NO_2, and SO_2 with similar lag patterns. Associations tended to be stronger in the cool season. The associations with NO_2 and SO_2 were the strongest in two-pollutant models. Patterns of association for pollutants with ischemic heart disease were similar in the two cities. The associations between O_3 and cardiac admissions were negative in London, but positive in Hong Kong. In conclusion, air pollution has remarkably similar associations with daily cardiorespiratory admissions in both cities, in spite of considerable differences between cities in social, lifestyle, and environmental factors.

Acute respiratory infections were the most common cause of illness and death in children in the developing world. Four studies showed an increase in infant mortality in relation to outdoor air pollution (Romieu, *et. al.*, 2002).

In Sao Paulo, Brazil, a study observed relationships between air pollutants and ischemic cardiac diseases such as angina and acute myocardial infarction in a representative cardiovascular centre emergency room (Lin, *et. al.,* 2003). Daily emergency room admissions to the Institute of the Heart, University of Sao Paulo, as well as data concerning daily air pollutant levels and meteorological variables, were collected from January 1994 to August 1995. Generalised additive Poisson regressions were fitted to the logarithm of the expected values of total emergency room visits due to angina or acute myocardial infarction, controlling for smoothing functions of season and weather and indicators for days of the week. All investigated pollutants were positively associated with

ischemic cardiovascular disease emergency room visits, and the time lags were relatively short. Only CO presented an effect that was statistically significant. An interquartile range increase in CO was associated with an increase of 6.4% (95% CI: 0.7-12.1) in daily angina or acute myocardial infarction emergency room visits.

2.3.4.2 Panel studies.

These studies are designed to elicit short term health effect data from exposure to ambient air pollutants. A panel of individuals is monitored for a predetermined period during which they record information such as pulmonary function, respiratory symptoms, occurrence of sore throats, colds wheezes, etc. Personal monitoring of exposure to pollutants may also be carried out. Advantages of panel methods are that (a) the panel can be selected to study vulnerable groups, such as asthmatics or the elderly and (b) that more subtle effects of air pollutants can be investigated i.e. respiratory effects such as increased coughing and wheezing that might not necessarily have resulted in a hospital admission. The studies are analysed in the same way as time series.

At Provo, Utah, a study evaluated changes in respiratory health associated with daily changes in fine particulate pollution (PM_{10}) (Pope, *et.al.*, 1991). Participants included a relatively healthy school-based sample of fourth and fifth grade elementary students and a sample of patients with asthma aged 8 to 72 years. Elevated PM_{10} pollution levels of 150 $\mu g\ m^{-3}$ were associated with an approximately 3 to 6% decline in lung function as measured by peak expiratory flow (PEF). Current day and daily lagged associations between PM_{10} levels and PEF were observed. Elevated levels of PM_{10} pollution also were associated with increases in reported symptoms of respiratory disease and use of asthma medication. Associations between degraded respiratory health and elevated PM_{10} pollution were observed even when PM_{10} levels were well below the 24-h national ambient air quality standard of 150 $\mu g\ m^{-3}$. Associations between elevated PM_{10} levels, reductions in PEF, and increases in symptoms of respiratory disease and asthma medication use remained statistically significant even when the only pollution episode that exceeded the standard was excluded. Concurrent measurements indicated that little or no strong particle acidity was present.

In a panel study carried out by Forsberg *et al.* (1993) 31 asthmatic patients from Pitea in northern Sweden were asked to complete an asthma diary, recording severe symptoms of shortness of breath, wheeze, cough and phlegm. Environmental conditions including

pollutant concentrations (sulphur dioxide, nitrogen dioxide and black smoke) as well as relative humidity and temperature were also monitored. Multivariate analyses of the diary data and environmental conditions indicated that "daily variations in the particulate pollution levels indicated by black smoke levels below the criteria limits, had significant effects on the risk of developing severe symptoms of shortness of breath". The overall conclusion for this study was of a association between black smoke levels and shortness of breath, and that these effects occur at lower smoke concentrations than previously suggested (Forsberg *et al.*, 1993).

A study carried out by Roemer *et al.* (1993) involved a panel of 73 children who had existing chronic respiratory symptoms. The children, aged between six to twelve years, were from two small non-industrial towns in Eastern Netherlands. A diary was kept by each child to record acute respiratory symptoms, and their peak lung flow was monitored twice per day. The study found that there was a small but statistically significant relationship between elevated PM_{10}, black smoke and SO_2 concentrations and reduced peak flow rates. There was also a positive relationship between these pollutants and the prevalence of wheezing. The study concluded that moderately elevated levels of these three pollutants results in mild to moderate effects in this susceptible group. No significant associations were found for nitrogen dioxide.
(Roemer, Hoek & Brunekreef, 1993)

A similar study carried out by (Ostro *et al.*, 1995) found that shortness of breath in asthmatic children aged seven to twelve years, from African-American backgrounds in Los Angeles was associated with elevated levels of ozone and particulate matter. The panel of 83 children monitored daily asthma symptoms, medication use, and peak flows over a period of three months. A follow-up study was carried out in 2001 (Ostro *et al.*, 2001).

During the winter of 1994, the relationship between daily changes in air pollution and respiratory health of children 7 to 12 years of age were studied in Kuopio, Finland (Timonen and Pekkanen, 1997). Seventy-four children with asthmatic symptoms and 95 children with cough only, living either in urban or suburban areas, were followed for 3 months. During the studied period, the mean daily concentration of particulate air pollution PM_{10} was 18 $\mu g\ m^{-3}$ in the urban area and 13 $\mu g\ m^{-3}$ in the suburban area. Lagged concentrations of PM_{10}, black smoke, and NO_2 were significantly associated with declines in morning peak expiratory flow (PEF) among asthmatic children. The regression coefficient (x10) for a 2-day lag of PM_{10} was -0.911 (SE, 0.386) in the urban and -1.05

(0.596), in the suburban area. Among children with a cough only, PM_{10}, black smoke, and NO_2 were not significantly associated with PEF. In the urban area, there was a significant association between SO_2, morning and evening PEF and incidence of upper respiratory symptoms among children who cough only. No other associations between air pollution and evening PEF or respiratory symptoms were observed. This study suggested that particulate air pollution was associated with respiratory health, especially among children with asthmatic symptoms.

A large-scale panel study conducted in the former Communist block countries of East Germany (Erfurt and Weimar) and Czechoslovakia (Sokolov) from 1990 to 1992 comprised 155 asthmatic children and 102 adults with a history of asthma (Peters *et al.*, 1996). Each panellist recorded details of daily symptoms, medication intake, and peak expiratory flow (PEF). Here sulphur dioxide concentrations and sulphate were found to be the best predictor of morbidity in children. The magnitude of he health effects was, however, relatively small, with a 128µg m^{-3} increase in sulphur dioxide over the previous five days resulting in a reduction of PEF of 0.90% and an increase in symptom rate of 14.7%.

Many of the previously discussed studies are from regions where pollution levels are only moderately elevated at certain times, e.g. winter. However, in a study based in Mexico City, where ambient standards are frequently exceeded, some strong associations were found in a panel study comprising 71 children (aged 5 to 7 years) with mild asthma (Romieu *et al.*, 1996). The measured variables were PEF and respiratory symptoms. It was found that a PM_{10} increase of 20 µg m^{-3} resulted in an 8% increase in respiratory illness on the same day. A increase in the weekly $PM_{2.5}$ mean of 10 µg m^{-3} resulted in an 20% increase in respiratory illness. Ozone was also indicated as a factor, with a 50 parts per billion (ppb) increase in ozone being associated with a 9% increase in respiratory illness.

Several other panel studies have supported the association between moderate health effects of elevated ozone and, particularly, PM_{10} levels in asthmatic children. A study 61 children, ages 7 to 13 years, of whom 77% were taking asthma medication showed that relatively low levels of ambient particulate pollution could result in increased medication rates in this vulnerable group (Gielen *et al.*, 1997). A study from Southern California where ozone concentrations were relatively high and PM_{10} relatively low found significant associations of asthma symptoms with both pollutants for a panel of 25 asthmatics aged 9

to 17 years (Delfino *et al.*, 1998). A study carried out in Paris on a panel of 84 asthmatic children was designed to address the issue of whether relatively low levels of ambient pollutants could exacerbate asthma (Segala *et al.*, 1998). A 50ug m-3 increase in sulphur dioxide levels was found to increase the odds of a (same day) asthma attack by 2.86; Morning peak expiratory flows (PEFs) were found to decrease by 5% at a lag of 3 days for the same sulphur dioxide decrease.

The relationship between outdoor air pollution and the respiratory health of children aged eight to ten years at the steel cities of New South Wales was investigated in Australia (Lewis et. al., 1998). The average annual outdoor air pollution for the nine areas was 18.6-43.7 $\mu g\ m^{-3}$ for PM_{10} and 1.6-9.0 ppb for SO_2. Pollutant concentrations were compared with health data that comprised the reported occurrence of four or more chest colds, four or more attacks of wheezing, and night-time cough without a cold for more than two weeks, all within the previous 12 months. The proportion of children reported to have the main outcome symptoms were: chest colds, 3.0%-9.7%; night cough, 12.3%-30.5%; and wheeze, 3.4%-11.3%. There was no significant association with SO_2, but a significant increase in the odds of symptoms per 10 $\mu g\ m^{-3}$ increase in PM_{10} on chest colds (odds ratio [OR], 1.43; 95% CI, 1.12-1.82) and night-time cough (OR, 1.34; 95% CI, 1.19-1.53), but not wheeze. Passive smoking was significantly associated with chest colds, but not with the other symptoms. Maternal allergy was associated with all three respiratory symptoms, but most strongly with wheeze. They concluded that their results provided evidence of health effects at lower than expected levels of outdoor air pollution in the Australian setting.

In a study carried out to determine whether severity of asthma is an indicator for susceptibility to air pollution, 60 non-smoking patients with intermittent to severe persistent asthma recorded their respiratory symptoms during the summer of 1995 (Hiltermann *et al.*, 1998). Whilst there was a consistent, positive association of the prevalence of shortness of breath with ozone, PM_{10}, black smoke and NO_2, and an association of bronchodilator use with both ozone and PM_{10} levels there was no evidence to support an association between severity of asthma and susceptibility to air pollution.

An important panel study carried out by (Pope *et al.*, 1999) was designed to look at mechanisms of mortality for particulate matter. The panel comprised 90 elderly people for whose blood oxygenation rates and pulse rates were measured during the winter of

1995/96 in Utah Valley. Blood oxygenation rates were not significantly associated with PM_{10}, however pulse rate did show an association with PM_{10} levels that had been elevated on the previous 1 to 5 days. The authors have suggested that the results indicate PM_{10} induced mortality may by through changes to the cardiac rhythm possibly as a result of pulmonary inflammation and cytokine release; the observed lags are consistent with this theory. A follow-up study by the same authors in 2004 (Pope *et al.*, 2004) looked at the effect of air pollution on (a) heart rate variability (HRV) and (b) blood markers for inflammation in a panel of 88 elderly panellists from communities in Utah. The study found small but statistically significant associations between $PM_{2.5}$ and both of the monitored health parameters, suggesting that exposure to this pollutant is one of the influencing factors.

The increasing realisation that ultrafine particulates (less than 0.1µm aerodynamic diameter) play a significant role in health impacts was investigated by reviewing a number of studies where ultrafine and fine (less than 2.5µm aerodynamic diameter) particulate concentrations had been monitored (Wichmann & Peters, 2000). In four panel studies involving asthmatic patients in Germany and Finland, decreased PEF was found to be associated with elevated particulate concentrations, in line with other studies discussed above, however the association *was* found to be strongest for ultrafine particulates. In a further study reviewed as part of this study, it was found that fine particulates were associated with more immediate effects, whereas ultrafine particulates showed a lag of up to four days; respiratory effects were more immediate for both fractions, whereas for cardiovascular effects the lag differences between the two fractions were more evident.

In a continuation of the more sophisticated recent panel studies, where blood biomarker levels have been monitored as a function of pollutant concentrations (ultrafine particulates), 57 male patients with coronary heart disease were studied during the winter of 2000/2001 (Ruckerl *et al.*, 2006). Blood biomarker levels were monitored for inflammation, endothelial dysfunction, and coagulation. Strong associations were found for inflammation markers and ultrafine particulates and PM_{10}, both immediately and for lags of two days. The other biomarker levels were not so consistent in their association, though there was evidence of some effect.

In summary, the panel studies have produced strong evidence of associations between respiratory and cardiopulmonary effects and pollutant concentrations, most particularly for particulate matter, but also for sulphur dioxide, ozone and sulphates. Many of the panel

studies have demonstrated respiratory effects at levels well below guideline values, particularly in vulnerable groups such as asthmatics. More recent investigations into different fractions of particulate matter have revealed that ultrafine and fine particulates tend to have stronger associations with health effects, both immediately and at lags of one to four days. Several recent panel studies have begun to look at biomarkers in addition to the usual diary and lung function records; these studies have the potential to elucidate mechanisms of toxicity in addition to determining associations. There is evidence that particulate matter causes inflammation of the airways and that the resulting cytokine release may affect pulse rate and cardiac rhythm.

2.3.4.3 Meta analysis

Essentially meta analysis is the statistical analysis of pooled individual studies, allowing the derivation of more robust health impact factors. There are two potential approaches: firstly the retrospective pooling and analysis of independent but comparable studies from the literature; and secondly the an approach where multicentre studies are planned in advance, with protocols governing all aspects of data collection and analysis, thus removing the inter-study variability inherent in the former approach. Meta analyses can be conducted on both panel and time series studies, though the latter are most prevalent.

One major European study was that of the Air Pollution on Health: European Approach (APHEA) project provided quantitative estimates of the short-term health effects of air pollution, using an extensive data base from 15 European cities in 10 different European countries with a total population exceeding 25 million (Katsouyanni, et al., 1995). Short-term effects of air pollution on mortality and hospital admissions were studied from several years' data of hospital admissions for all respiratory causes in five West European cities (London, Amsterdam, Rotterdam, Paris, Milan) (Spix et.al., 1988). The most consistent and strongest finding was a significant increase of daily admissions for respiratory diseases (adults and elderly) with elevated levels of ozone. It was stronger in the elderly, had a rather immediate effect (same or next day), and was homogeneous over cities. During the warm season, the elderly were affected more. From the available data in all cities, sulphur dioxide was not associated consistently with an adverse effect. The effect of Black Smoke was significantly stronger with high nitrogen dioxide levels on the same day, but nitrogen dioxide itself was not associated with admissions.

Investigation of the short-term effects of air pollution on hospital admissions for chronic obstructive pulmonary disease (COPD) in Europe was made as part of APHEA (Anderson, et. al., 1997). Using a standardised approach to data eligibility and statistical analysis, data was analysed from the cities of Amsterdam, Barcelona, London, Milan, Paris and Rotterdam. For all ages, the relative risks (95% confidence limits (95% CL)) for a 50 $\mu g\ m^{-3}$ increased in daily mean level of pollutant (lagged 1-3 days) were (for 95% CL): SO_2 1.02 (0.98, 1.06); black smoke 1.04 (1.01, 1.06); TSP 1.02 (1.00, 1.05), NO_2 1.02 (1.00, 1.05) and O_3 (8 h) 1.04 (1.02, 1.07). The results confirmed that air pollution was associated with daily admissions for chronic obstructive pulmonary disease in European cities with widely varying climates.

Again, as part of the APHEA project, the association between airborne particles and hospital admissions for cardiac causes was examined in eight European cities (Barcelona, Birmingham, London, Milan, the Netherlands, Paris, Rome, and Stockholm) (Le Tertre, et. al., 2002). In each city, autoregressive Poisson models were used. Independent factors were long term trends, seasons, influenza epidemics, and meteorology to assess the short-term effects of particles. Other pollutants were also examined. City specific results were pooled in a second stage regression to obtain more stable estimates and examine the sources of heterogeneity. The pooled percentage increases associated with a 10 $\mu g\ m^{-3}$ increase in PM_{10} and black smoke were respectively 0.5% (95% CI: 0.2 to 0.8) and 1.1% (95% CI: 0.4 to 1.8) for cardiac admissions of all ages, 0.7% (95% CI: 0.4 to 1.0) and 1.3% (95% CI: 0.4 to 2.2) for cardiac admissions over 65 years, and, 0.8% (95% CI: 0.3 to 1.2) and 1.1% (95% CI: 0.7 to 1.5) for ischaemic heart disease over 65 years. The effect of black smoke remained practically unchanged at reduced concentrations of CO and only somewhat reduced for lower NO_2 concentrations. Results for ischaemic heart disease below 65 years and for frequency of strokes for people over 65 years were inconclusive.

In a North American study, Boadway (1998) has reviewed the evidence of the health effects of air pollutants on the basis of seven key Ontario studies (Boadway et. al., 1998). These findings were highly significant for people living in the Great Lakes basin (and particularly the Windsor-Quebec corridor), where high levels of certain air pollutants (e.g., ground-level ozone and ultra-fine particles) occurred more frequently than in other parts of Canada. The findings of the research included the following: in a Toronto study, a 2% to 4% excess of respiratory deaths were attributable to pollutant levels; children living in rural Ontario communities with the highest level of airborne acids were significantly more likely

to report at least one episode of bronchitis, as well as to show decreases in lung function; and have been linked to increases in pollutants, emergency room visits and hospitalisations in Ontario.

More recently two large European and North American studies have produced some remarkably consistent results on air pollutant health effects (Brook *et al.*, 2004); the first an extension to the original APHEA project (APHEA-2), based on time series analysis on 43 million people in 23 European cities (Katsouyanni *et al.*, 2001), and the second, NMMAPS, based on health outcomes from 50 million people in the 20 largest cities in the United States in the United States (Dominici *et al.*, 2003)

NMMAPS found that for each 10μg m^{-3} increase in PM_{10} concentration, the next-day mortality rate increased by 0.21% for all causes, and 0.31% for specific cardiopulmonary mortality. In the APHEA study, a stronger association was observed, with total mortality and cardiovascular deaths increasing by 0.6% and 0.69%, respectively for the same PM_{10} increment. There is also evidence from the APHEA-2 study of longer lag effects, for example a 1.92% increase in cardiovascular mortality over a 40 day lag period. A further interesting finding from the APHEA-2 study was ability of nitrogen dioxide concentrations to increase the strength of association between PM_{10} and mortality rates, though this was not observed in the NMMAPS study. Sub-mortality observations show significant increases in admission rates of 0.8% and 0.7% for heart failure and ischemic heart disease, respectively, for each. A new APHENA (Air Pollution and Health: A European and North American Approach) project is attempting to bring together the European and North American studies (Samet, Katsouyanni & Principal Investigators, 2006). Recently, the study has reported on the effects of PM10 which showed that for all causes of mortality, across all ages and cities a range of 0.2% to 0.6% increase in daily mortality is observed for a 10μg m^{-3} increase in PM_{10} concentration (Samoli *et al.*, 2008).

In Asia, where PM_{10} levels are generally higher than those encountered in the Western studies, the first meta analysis was not published until 2004 by the Health Effects Institute, Boston (HEI, 2004). The study, based on 100 peer-reviewed time series analyses from nine different countries showed similar health impacts to the NMMAPS and APHEA-2 studies, with a 0.49% increase in mortality rate per 10μg m^{-3} increment in PM_{10} concentration. The study of health impacts of air pollutants in Asia is now coordinated by the Public Health and Air Pollution in Asia (PAPA) project (Wong *et al.*, 2008). The first phase of the PAPA study was carried out using data from Bangkok, Thailand,

from 1999 to 2003, Hong Kong, China, from 1996 to 2002, and Shanghai and Wuhan, China, both from 2001 to 2004. A common protocol is available for both data collection and analysis. This four-city meta-analysis revealed that health impacts are as high (PM_{10}) or higher (NO_2 and SO_2) than observed in the Western studies. For example, increases in daily mortality rates per 10μg m^{-3} increment in pollutant concentration were 2–3 times greater for NO_2 than those derived from the APHEA-2 project (1.23% vs. 0.3%). For SO_2, the increased mortality rate (1.00%) was twice that reported in the HEI study (HEI, 2004) For PM10, the increased mortality rate of 0.55% for all natural causes of death at all ages was comparable to the 0.49% from all Asian cities (HEI 2004), 0.5% from NMMAPS (Samet *et al.* 2000), and 0.6% from the APHEA project (Anderson *et al.* 2004). For ozone, the increased mortality of 0.38% per 10μg m^{-3} increment was higher than observed for APHEA-2 (0.2%) and NMMAPS (0.26%).

2.3.4.3 Long-term studies

Two principal methods used is assessing chronic effects of air pollution are cohort studies and 'air pollution interventions'. Cohort studies require the recording of health end-points in a large group of people at specific intervals, often ten or more years. The cohorts are well characterised in terms of smoking status, occupations, body weight etc. so that confounding factors can be removed from the analysis. In order to estimate the health impacts, comparisons are made between two or more cohorts exposed to different pollutant concentrations. Mortality is the usual end-point that is measured, allowing these studies to give an estimate of the average reduction in life expectancy as a result of exposure to pollution. Intervention studies look at pollution related health end-points before and after the introduction of measures to reduce pollution levels; these may either be regulatory interventions or may arise as the result of other factors, for example closure of a particularly polluting industry.

Perhaps the most famous cohort study is the Harvard Six Cities study, which recorded mortality rates in 8111 adults from six U.S. cities at fourteen to sixteen year follow-ups (Dockery *et al.*, 1993); initial enrolment into the study was in the mid-1970s. After adjusting for smoking (the most significant determinand of mortality) and other risk factors, there was a statistically significant association between mortality and air pollution, most strongly with the fine particulate fraction. The adjusted mortality-rate ratio for the most polluted of the cities as compared with the least polluted was 1.26 (95 percent confidence interval, 1.08 to 1.47), that is to say that there is a 26% increase in mortality rates as a result of

exposure to air pollution between the most and least polluted cities. Lung cancer and cardiopulmonary disease were positively associated with air pollution.

An extended follow up to the Six Cities study, over a time period when pollution levels have fallen as a result of regulatory interventions, has recently been published (Laden *et al.*, 2006). This study confirmed the association between lung cancer and cardiovascular mortality as a result of exposure to air pollution, with the strongest association with $PM_{2.5}$. In another significant study, a cohort of 552,138 adults from 151 U.S. metropolitan areas was enrolled and mortality was monitored through to 1990 (Pope *et al.*, 1995). Mortality risk factors as a result of exposure to air pollution, after adjustment for confounding factors such as smoking, were calculated from multivariate analysis. An association was found between exposure to particulate and sulphate air pollution and mortality, specifically cardiovascular and lung cancer. The adjusted relative risk factor between the most and least polluted areas was 1.15 (CI, 1.09 to 1.22). This study was followed up in a much larger cohort study that enrolled 1.2 million adults from metropolitan areas throughout the United States; mortality was followed through to 1999 (Pope *et al.*, 2002). The study concluded that fine particulate and sulphur oxide-related pollution were associated with all-cause, lung cancer, and cardiopulmonary mortality, with each 10 $\mu g\ m^{-3}$ increment in fine particulate air pollution being associated with approximately a 4%, 6%, and 8% increased risk of all-cause, cardiopulmonary, and lung cancer mortality, respectively.

An important within-city cohort study designed to assess the impact of traffic related pollution was carried out in the Netherlands (Hoek *et al.*, 2002). The cohort of 5000 adults was followed for 8 years and found that mortality was more strongly linked to traffic pollution than to city-wide background concentrations. For example, people living near a major road had a relative risk factor of cardiopulmonary mortality 1.95, (95% CI 1.09 to 3.52) compared to those who did not. This study highlighted the importance of traffic emissions in determining mortality rates.

The most famous of the intervention studies comes from the Salt Lake, and Cache valleys, in Utah (Pope, 1991). The study assessed the association between respiratory hospital admissions and PM_{10} pollution during April 1985 through March 1989. Utah and Salt Lake valleys had high levels of PM_{10} pollution that violated both the annual and 24-h standards issued by the EPA. In the Cache Valley, much lower PM_{10} levels occurred. Utah Valley experienced the intermittent operation of its primary source of PM_{10} pollution, an integrated steel mill. Bronchitis and asthma admissions for preschool-age children were

approximately twice as frequent in Utah Valley when the steel mill was operating, compared to periods when it was not. Similar differences were not observed in Salt Lake or Cache valleys. Even though Cache Valley had higher smoking rates and lower temperatures in winter than did Utah Valley, per capita bronchitis and asthma admissions for all ages were approximately twice as high in Utah Valley. During the period when the steel mill was closed, differences in per capita admissions between Utah and Cache valleys narrowed considerably. Regression analysis also demonstrated a statistical association between respiratory hospital admissions and PM_{10} pollution. The results suggest that PM_{10} pollution plays a role in the incidence and severity of respiratory disease

Table 2.5 summarises the main findings of the studies discussed above.

Table 2.8 Global Summary of Health Effect

Pollutant	Area/City	Effect	Risk Factor	Reference/ Country
O_3	London, Amsterdam, Rotterdam, Paris, Milan,	respiratory admissions	4% increase in admission for a 50 $\mu g\,m^{-3}$ increased in daily mean level of pollutant	Spix ,et,al 1988 APHEA
Black Smoke	London, Birmingham, Amsterdam, Rotterdam, Paris, Milan	respiratory admissions	4% increase in admission for a 50 $\mu g\,m^{-3}$ increased in daily mean level of pollutant	Andersen, et, al 1997 APHEA
	Barcelona, Birmingham, London, Milan, the Netherlands, Paris, Rome, and Stockholm.	cardiac admissions	1.1% increase in admission for a 10 $\mu g\,m^{-3}$ increased in daily mean level of pollutant	Le Tertre, et, al 1997 APHEA
PM_{10}	Barcelona, Birmingham, London, Milan, the Netherlands, Paris, Rome, and Stockholm	cardiac admissions	0.5 % increase in admission for a 10 $\mu g\,m^{-3}$ increased in daily mean level of pollutant	Andersen, et, al 1997 APHEA
	Thailand	Cardiovascular mortality	1-2% increase in cardiovascular mortality for a 10 $\mu g\,m^{-3}$ increased in daily mean level of pollutant	Ostro, et, all 1999 Thailand
	Birmingham,	Respiratory admissions	2.4 % increase in admission for a 10 $\mu g\,m^{-3}$ increased in daily mean level of pollutant	Waltus, et, all 1994 UK
	London	respiratory admissions	5.7 % adult aged 15-64; 10.2 % adult aged 65 and over increase in admissions for a 16-47 $\mu g\,m^{-3}$ increased in daily mean level of pollutant	Hajat, et, al 2002 UK.
	steel cities of New South Wales	respiratory admissions	chest colds, 3.0% - 9.7%; night cough, 12.3% - 30.5%; and wheeze , 3.4%-11.3% increase	Luvis , et, al 1998 Australia
	Thailand	respiratory mortality	3-6% increase in respiratory mortality for a 10 $\mu g\,m^{-3}$ increased in daily mean level of pollutant	Ostro, et, all 1999 Thailand
	Birmingham	cerebrovascular admissions	2.1% increase in admissions for a 10 $\mu g\,m^{-3}$ increased in daily mean level of pollutant	Wong, et, al 2002 U.K
TSP	London, Amsterdam, Rotterdam, Paris, Milan	respiratory admissions	2% increase in admissions for a 50 $\mu g\,m^{-3}$ increased in daily mean level of pollutant	Anderson, et, al 1997 APHEA
SO_2	London	respiratory admissions	3.5% childhood increase in admissions for a 13-31 $\mu g\,m^{-3}$ increased in daily mean level of pollutant	Hajat, et, al 2002 U.K
NO_2	London, Amsterdam, Rotterdam, Paris , Milan	respiratory admissions	2% increase in admissions for a 50 $\mu g\,m^{-3}$ increased in daily mean level of pollutant	Spix, et, al 1988 APHEA
CO	Sao Paulo	Daily angina or acute myocardial infarction emergency room visits	6.4% increase in admissions for interquartile range increase in CO	Lin, et, al 2003 Brazil

2.3.4.13 Review of the Multiple Regression Analysis and Human Health Concern

This section reviews studies that have been carried out on the application of multiple regression analysis methods (MLRs) to determine correlations between air pollution and various human health effects.

Multiple linear regression was used to analyse relationships among community air quality, socio-economic variables, and mortality rates for all cancers, respiratory system cancer, respiratory disease, and external causes, for U.S. cities during 1969-1971. Most air pollution variables were not found to be significant, except the trace metal manganese, which was associated with cancers and respiratory disease. In this study, low ambient concentration were observed, so it is likely that manganese is serving as a surrogate for some other effect, such as occupational influences (Lipfert, 1980).

In the city of Tver (Russian Federation), multiple regression technique was used to prove a relationship between high temperatures, air pollution, and morbidity and mortality, and estimate the power of association between environmental factors and public health. The four warmest months were analysed (June to September) for three years (1999, 2001 and 2002). The result found a positive correlation at the 95% level of statistical significance. An increase of 10°C in the daily maximum temperature caused one extra mortality. Of all types of mortality, only two diagnoses (cerebrovascular diseases and Poisoning and suicide) were shown to be positively correlated with temperature. Average daily concentrations SO_2, NO_2, NO, TSP, CO, CO_2 were found to be positively correlated with maximum daily temperature: TSP, SO_2, NO_2 and NO. The correlation Equations were:

$TSP = 1.38 + (0.06+ -0.02)T$ Equation 2.19

$SO_2 = -0.9 + (0.49+ -0.24)T$ Equation 2.20

$NO_2 = -1.61 + (0.19+ -0.04)T$ Equation 2.21

$NO = -0.54 + (0.13+ -.03)T$ Equation 2.22

Individually small associations were observed for PM_{10}, SO_2, NO_2, NO, NO_x, CO and benzene but not meteorological factors (temperature and rainfall) or O_3 with asthma emergency department admissions. The concentration of benzene was the only variable associated independently with emergency-department asthma admissions in children. Thus, benzene was a more reliable method of measuring exposure to vehicle exhaust emissions than other pollutants (Shields, 2003).

In Hong Kong, analyses were carried out using a times series analysis approach during 1994 and 1995. Significant correlations were found between pollutants (NO_2, SO_2, O_3 and inhalable suspended particulates) and hospital admissions (respiratory diseases, cardiovascular diseases, and bronchial asthma), between O_3 and deaths (respiratory and circulatory diseases), and between NO_2 and respiratory mortalities. For every 100 μg m^{-3} increase of pollutant level, a 13 to 40% and 13 to 18% increase in hospital admissions for respiratory diseases and for cardiovascular diseases, respectively. The relative risks of deaths from respiratory and circulatory diseases for an increase in 100 μg m^{-3} of O_3 were 2.62 and 1.27 respectively, while that of respiratory mortalities for NO_2 was 1.31. The Poisson regression model was validated using data for the first half-year of 1996. The predicted daily hospital admissions were found to fit the observed pattern fairly well. A reduction of 100 μg m^{-3} of ambient O_3 concentration would result in a 28% fall in hospital admissions for respiratory diseases and a 38% fall in respiratory mortalities. A similar reduction in NO_2 would lead to a 29% fall in hospital admissions for respiratory diseases and a 24% fall in respiratory mortalities (Wai *et al*, 1997).

Another study analysed the short-term association between air pollutants (SO_2, PM_{10}, NO_2 and O_3) and asthma emergency room admissions in Madrid, Spain, in 1995-1998, adjusting for four types of pollen with allergenic potential (Olea europaea, Plantago sp., Poaceae and Urticaceae). Data were analysed using autoregressive Poisson regression and generalised additive models (GAM). The strongest associations were observed at one day lag for O_3, and three days lag for the remaining pollutants. The model showed that a 10 μg m^{-3} rise in pollutant level led to relative risks of: 1.039 for SO_2; 1.033 for NO_2; and 1.045 for O_3. Adjustment for the different types of pollen led to no substantial variation in these associations. In the multipollutant models for cold-season pollutants (including PM_{10}, SO_2 and the four types of pollen) and photochemical pollutants (including NO_2, O_3, and the four types of pollen) the associations for PM_{10}, NO_2 and O_3 held, but no relationship with SO_2 was evident. GAM analysis yielded the same results, both in terms of lags and of quantification of the effect for all pollutants. In conclusion, the usual air pollution levels in Madrid were associated with an increase in asthma emergency room admissions, and this association remained controlling for the presence of ambient pollen (Galan, 2003).

The independent effect of environmental variables on daily counts of death from myocardial infarction in a subtropical region in Sao Paulo in Brazil has been studied by Sharovsky *et al* (2004). Pearson correlation coefficients was used to investigate

associations between daily deaths (myocardial infarction 12,007 fatal events from 1996 to 1998), weather variables (temperature, humidity and barometric pressure), and pollutant concentration (sulphur dioxide, carbon monoxide, and inhalable particulate). The model was adjusted in a linear fashion for relative humidity and day-of-week, while nonparametric smoothing factors were used for seasonal trend and temperature. It was found that a significant association of daily temperature with deaths due to myocardial infarction ($P < 0.001$), with the lowest mortality being observed at temperatures between 21.6 and 22.6°C. Relative humidity appeared to exert a protective effect and SO_2 concentrations correlated linearly with myocardial infarction deaths, increasing the number of fatal events by 3.4% (relative risk of 1.03; 95% confidence interval = 1.02-1.05) for each 10 μg m^{-3} increase. In conclusion, this study provides evidence of important associations between daily temperature and air pollution and mortality from myocardial infarction in a subtropical region, even after a comprehensive control for confounding factors (Sharovsky *et al.*, 2004).

In Arizona, a study focused on the associations between lung function measurement (respirometry and peak expiratory flow liability) and air pollution (suspended SO_4^{2-}, SO_2, O_3, and indoor particles). Non-smokers, 1,391 people since 1977, were selected as samples. PM_{10} exceeded 100 μg m^{-3} on 54 d/yr, and was associated with a 7.2% decrement in one-second forced expiratory volume (FEV1), in males whose parents had asthma, bronchitis, emphysema, or hay fever and with increased peak expiratory flow liability of 0.8% for all females and 0.6% for all males. An increase in mean SO_4^{2-} concentration of 1.6 μg m^{-3} was associated with a 1.5% decrement in FEV1, as percent of predicted, in all males. An increase of 23 ppb of O_3 as an 8-h average was associated with a 6.3% decrement in FEV 1 in males whose parents has asthma, bronchitis, emphysema or hay fever (Abbey *et al.*, 1998).

In California, multiple regression techniques were used in place of laboratory experiments to control for other factors. One study used biweekly data from 117 standard metropolitan areas to derive the following equation:

$$Y = 19.61 + \underset{[2.5]}{0.041x_1} + \underset{[3.2]}{0.71x_2} + \underset{[1.7]}{0.001x_3} + \underset{[5.8]}{0.41x_4} + \underset{[18.9]}{6.87x_5} \qquad \text{Equation 2.23}$$

Where Y = mortality rate, per 10,000 population (average = 91.4)

X_1 = average suspended particulate reading (average = 118.1)

X_2 = minimum sulphate reading (average = 4.7)

X_3 = population density per square mile (average = 756.2)

X_4 = fraction of population that is non-white (average = 12.5)

X_5 = fraction of population 65 and older (average = 8.4)

[ ] = t-values

For the average city, a 10 percent increase in the average suspended particulate and minimum sulphate increased the mortality rate by 0.48 and 0.34, respectively. A change 10 μg m^{-3} in daily PM_{10} (about 9 percent) averaged over three days was associated with a 1.1 percent increase in mortality (95 percent confidence interval: 0.6 to 1.5 percent). Death from respiratory and cardiovascular disease was more responsive to changes in PM_{10} than total mortality was. The same holds for mortality among men and mortality among individuals older than 64 (Economics Department, 2005).

From a slightly different perspective, one study has investigated the potential financial benefits from pollution reduction in terms of decreased use of medical care. With medical care spending exceeding one trillion U.S. dollars per year, even a reduction of only a few percent would save society tens of billions of dollars annually. Three previous studies of the relationship between medical care use and pollution have examined only a limited number of illnesses and have been based on a limited number of areas. A few have focused specifically on children or the elderly, but with relatively small samples (Fuchs and Frank, 2002).

A study by Higgens (2000) proved that O_3 and NO_2 could enhance the effect of allergen in sensitised asthmatic subjects under natural exposure conditions. Previous laboratory-based studies reported that these pollutants can potentiate but thus far there had been no research under natural conditions. The research involved thirty-five subjects with clinical diagnoses of asthma or chronic obstructive pulmonary disease who kept peak expiratory flow (PEF) records for a 4-week period during late summer, with concurrent measurement of spore and pollen counts and pollution levels. The effect on PEF of aeroallergen, and of the interaction between aeroallergen and pollutant levels was determined by multiple regression analysis. Interaction between total spore count and ozone was statistically significant. A decreasing value of mean PEF was associated with increasing spore count (same-day and 24-h lag level), with the effects greater for higher prior ozone concentrations. It could be concluded that ozone can potentiate the effect of aeroallergens in subjects with bronchial hyper reactivity under natural exposure conditions (Higgins B.G., 2000).

In Chile, pneumonia and respiratory illness were the major causes of death for children ages one to four (INE, 1999). Air pollution in this area was trapped because of the geography, which forms a closed basin of hills, leading to increasing in contaminant concentrations (Sandoval, 1993). The atmospheric characteristics also contributed towards low dispersion, particularly during the coastal low that formed when skies are clear together with low humidity was low, high surface temperatures, and weak easterly winds. The coastal low propagates southward, eventually bringing cold humid maritime air into the River Maipo Valley (Ulriksen, 1993). The study identified weather patterns and air pollution regimes most highly associated with increased mortality. Stepwise multiple regression was used to determine the relationships between weather and pollution variables in explaining the variation in excess deaths. It could be concluded that NO_2 had a significant effect on disease-related mortality after controlling for the effect of weather. The results suggested that during the winter in Santiago, the effects of weather and pollution on mortality were additive and independent (Bart *et al.*, 1995).

2.4 Dispersion Modelling

This section reviews the main factors that influence the dispersion of pollutants in the atmosphere, and the techniques that can be used for predicting air pollutant concentrations.

2.4.1 Diurnal Variation of the Boundary Layer

Although the general dependencies between meteorological parameters and urban air quality are well known, there is no common quantitative rule of relative importance and impact of a particular meteorological parameter on the air quality in the specific area of interest. During recent years, it has been demonstrated that area specific infrastructure, land use and microclimatological conditions create specific air pollution characteristics.

Diurnal variation of the lower part of the atmosphere, called the planetary boundary layer (BL) where air pollution phenomena are observed, is studied. The BL is defined as a part of the troposphere that is directly influenced by the presence of the Earth's surface and responds to surface forcing with a time scale of about an hour or less (Stull, 1994). Over land surfaces in high pressure regions, the boundary layer has a well defined structure that has a diurnal variation cycle (Fig. 2.21) The mixing properties and diurnal evolution of the of BL will determine whether the emitted pollutants are effectively dispersed and transported away from the emission source or will accumulate at the emission source location reaching high air pollutant concentration levels.

The surface layer (SL) is the region at the bottom of the BL, where turbulent fluxes and stress vary by less than 10% of their magnitude. During the day SL height varies from several cm at night to about 300 metres in the midday (Stull, 1994). In the SL, turbulence is generated in a mechanical way, caused by friction and wind, and by buoyancy, i.e. vertical motions caused by air density differences in the Earth's gravity field. The driving force for mechanically generated turbulence is vertical flux density of momentum, whereas heating drives buoyancy at the surface. When heat flux is directed downward, buoyancy is suppressing turbulence, and the only source for turbulence is a mechanical one. The vertical transfer of heat and momentum in the SL over a flat, horizontally uniform terrain is described well by the Monin-Obukhov similarity theory (Stull, 1994).

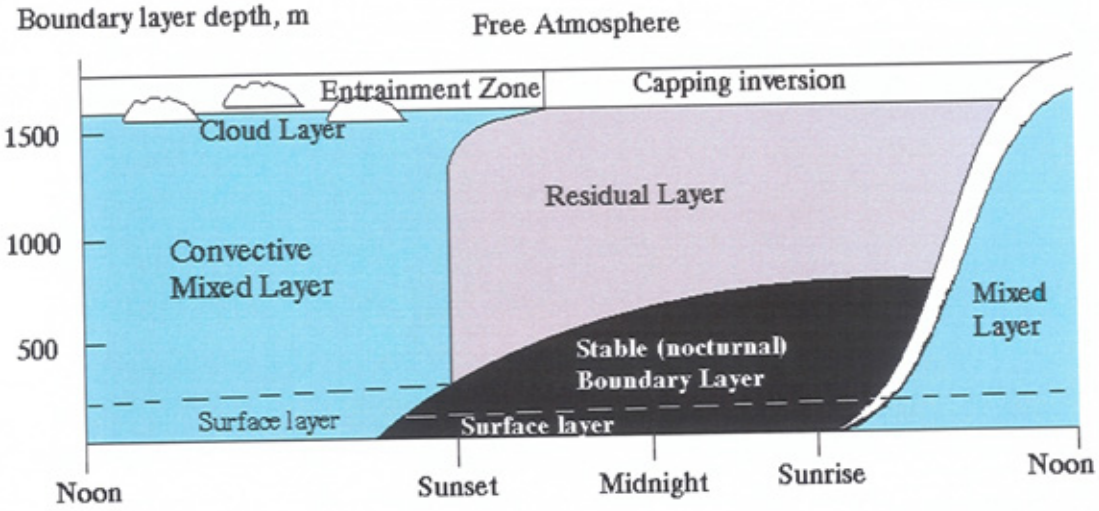

Figure 2.21 Diurnal variation of the boundary layer structure.
Source: Stull (1994)

The basic form of the relationship between air pollution concentrations and meteorological parameters can be derived using the so-called box model (Derwent, 1995; Zannetti, 1992) (Fig. 2.22). According to the box model, the pollutant emissions are confined to a mixing height of H [m], which exhibits a daily variation. Within the mixing layer, the unidirectional wind speed, W [m/s] carries a background concentration C_0 [g m^{-3}] into a source area. The source area itself has an emission density Q [g m^{-2}, s] as well as the source length in the direction of the wind vector, L [m]. Assuming that the local emissions are mixed instantaneously, the concentration, C [g m^{-3}], averaged over the source region can be estimated by the expression:

$$C = C_0 + \frac{QL}{WH} \qquad \text{Equation 2.24}$$

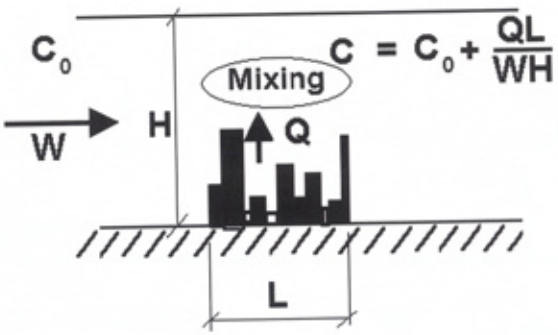

Figure 2.22 Simple meteorological box model.

The total concentration, C, is the sum of background and the locally contributed values. The second term on the right side represents the local contribution and is proportional to the source strength (Q*L) and inversely proportional to the ventilation coefficient (W*H). According to the box model, the dependence of the local contribution on the wind speed and the mixing height is inverse. The concentration, C, is the highest at low wind speeds, because the pollutants accumulate due to poor ventilation. With increasing wind speed, concentrations asymptotically approach the regional background concentrations due to the rapid dilution of local contributions

Although in the literature, the importance of the wind speed on the horizontal pollutant dispersion is generally acknowledged, the quantitative estimation of its effects on the air quality at the particular area has to be determined separately. The strength of correlation and form of the relationship reflects the receptor distance to the emission source, surface topography and roughness.

2.4.2 Atmospheric Pollutants of Interest

Carbon monoxide is produced by the incomplete combustion of fossil fuels and in urban areas its major source is exhaust from light duty motor vehicles. The dominant sink process for CO is the reaction with the hydroxyl radical (OH). The lifetime of the CO molecule is about two months, therefore in relatively short timeframe used for air pollution modelling it is considered to be non-reactive. Other CO removal mechanisms that transport and dispersion can be neglected (Hanna, 1978). The temporal trend of the ground level CO concentration is supposed to be determined by its emission rate and dispersive properties of the atmospheric surface layer.

Nitrogen oxides and ozone take part in the chemical transformations in the urban atmosphere; therefore, it is necessary to examine their dynamics together. Nitrogen oxides NOx (NOx = NO + NO_2, [ppb]) are released into the atmosphere mainly in the form of nitric oxide as a product of the reaction of N_2 with O_2 during the high temperature combustion processes of the fossil fuel released into the atmosphere mainly in the form of nitric oxide as a product of the reaction of N_2 with O_2 during the high temperature combustion processes of the fossil fuel

$$N_2 + O_2 \rightarrow 2NO \qquad \text{Equation 2.25}$$

It is established that approximately 90% of the emitted NOx consists of NO gas. The emissions of nitrogen oxides greatly increase with combustion temperature. Once emitted into the atmosphere, nitric oxide is relatively rapidly oxidized to nitrogen dioxide by reaction with ozone:

$$NO + O_3 \rightarrow NO_2 + O_2 \qquad \text{Equation 2.26}$$

Under the influence of intense sunlight nitrogen dioxide decays again into nitrogen monoxide and oxygen atoms, which react with O_2 to form ozone again:

$$NO_2 + h\nu\ (\lambda < 430nm) \rightarrow NO + O \qquad \text{Equation 2.27}$$

$$O + O_2 \rightarrow O_3 \qquad \text{Equation 2.28}$$

The photolysis of nitrogen dioxide does not give any net production of ozone, since ozone and nitric oxide react to form nitrogen dioxide. For the net formation of ozone, there must be a competing reaction of oxidizing NO to NO_2 that requires intense sunshine and high temperatures (>25 ° C). Consequently, high levels of ozone are generally observed under sunny, summertime conditions at locations where the air mass has previously collected emissions of hydrocarbons and nitrogen oxides. Due to reaction 2.26, significant concentrations of O_3 and NO cannot co-exist and therefore urban areas display, on the average, a slightly lower ozone concentration in the near-ground air than rural regions. Nitrogen oxides have a lifetime of approximately one day with respect to conversion to nitric acid. Nitric acid is removed from the atmosphere by direct deposition on the ground or transfer to aqueous droplets (e.g. cloud or rainwater), thereby contributing to acid deposition (Finlayson-Pitts *et al.*, 1986). The main chemical sink for ozone in a polluted area is the reaction between nitric oxide and ozone, atmospheric decomposition and surface deposition.

The major source of SO_2 is the combustion of sulphur-containing fuels. The rate of SO_2 emission can be estimated by the rate of fuel consumption, the percent of sulphur in the fuel (fuel contains up to 0.3% of sulphur) and the fact that 90% or more sulphur is emitted in the form of SO_2. The intensity of SO_2 emission depends essentially on the ambient temperature since when the temperature decreases, combustion of the heating systems and consequently SO_2 emissions increases. The ambient temperature factor is the most

important to the SO_2 concentration levels in the atmosphere, although it has very slight physical influence on the diffusion conditions.

2.4.3 Air Quality Models

Air quality models use numerical and mathematical techniques to simulate physical and chemical processes of air pollutants that disperse in the atmosphere. Simulation of the models are based on inputted data, including meteorological data, geographical data, and emission data. Some models analyse primary pollutants or physical process of pollutants that emitted directly from sources. Secondary pollutants need more complex modelling to simulate chemical process. The results can assist in strategy planning and air quality management systems designed to protect human health, environment, and property. The air quality models that are widely used are dispersion models, photochemical models and receptor models. The dispersion model can estimate the concentration of pollutants at specified ground-level receptors surrounding an emissions source. This model type is the most commonly used to assess environment effects of pollution sources. The photochemical models are used in regulatory or policy assessments to simulate the impacts from all sources by estimating pollutant concentrations and deposition of both inert and chemically reactive pollutants over large spatial scales. The receptor models use observational techniques which use the chemical and physical characteristics of gases and particles measured at source and receptor to both identify the presence of and to quantify source contributions to receptor concentrations.

The USEPA classifies models in terms of: referred/recommended model, alternative model, screening tool and related programs. Referred/recommended models are required to be used for State Implementation Plan revisions for existing sources and for New Source Review and Prevention of Significant Deterioration programs. There are two main recommended models: AERMOD and CALPUFF (US EPA, 2007). The AERMOD modelling system is a steady-state plume model that incorporates air dispersion based on planetary boundary layer turbulence structure and scaling concepts, including treatment of both surface and elevated sources, and both simple and complex terrain. The CALPUFF modelling system is a non-steady-state puff dispersion model that simulates the effects of time- and space-varying meteorological conditions on pollution transport, transformation, and removal. CALPUFF can be applied for long-range transport and for complex terrain.

The alternative models can be used in regulatory applications with case-by-case justification to the reviewing authority. The alternative models include ADAM, ADMS-3, AFTOX, ASPEN, DEGADIS, HGSYSTEM, HOTMAC/RAPTAD, HYROAD, ISC3, ISC-PRIME, OBODM, OZIPR, Panache, PLUVUEII, SCIPUFF, SDM, and SLAB (US EPA, 2007)

Screening Tools are models that are applied before applying a refined air quality model to determine if refined modelling is needed. The screening models are the following: AERSCREEN, COMPLEX1, CTSCREEN, RTDM3.2, SCREEN3, TSCREEN, VALLEY, and VISCREEN (US EPA, 2007)

The related programs are supporting programs for some of the preferred/recommended, alternative, and screening models. The models include AERMAP, AERSURFACE, BPIP, BPIPPRM, CALMPRO, CHAVG, CONCOR, and EMS-HAP (USEPA, 2006).

The dispersion models are based on the same basic principal of the complete equation for Gaussian dispersion modelling of continuous, buoyant air pollution plumes (from Beychok, 2005):

$$C = \frac{Q}{u} \cdot \frac{f}{\sigma_y \sqrt{2\pi}} \cdot \frac{g_1 + g_2 + g_3}{\sigma_z \sqrt{2\pi}}$$

where: f = crosswind dispersion parameter = $\exp\left[-y^2/(2\,\sigma_y^2)\right]$

g = vertical dispersion parameter = $g_1 + g_2 + g_3$

g_1 = vertical dispersion with no reflections = $\exp\left[-(z-H)^2/(2\,\sigma_z^2)\right]$

g_2 = vertical dispersion for reflection from the ground
= $\exp\left[-(z+H)^2/(2\,\sigma_z^2)\right]$

g_3 = vertical dispersion for reflection from an inversion aloft

$$= \sum_{m=1}^{\infty} \{ \exp\left[-(z-H-2mL)^2/(2\,\sigma_z^2)\right] + \exp\left[-(z+H+2mL)^2/(2\,\sigma_z^2)\right] + \exp\left[-(z+H-2mL)^2/(2\,\sigma_z^2)\right] + \exp\left[-(z-H+2mL)^2/(2\,\sigma_z^2)\right]$$

C = concentration of emissions, in g m^{-3}, at any receptor located:
x metres downwind from the emission source point
y metres crosswind from the emission plume centreline
z metres above ground level
Q = source pollutant emission rate, in g/s
u = horizontal wind velocity along the plume centreline, m/s
H = height of emission plume centreline above ground level, in m
σ_z = vertical standard deviation of the emission distribution, in m
σ_y = horizontal standard deviation of the emission distribution, in m
L = height from ground level to bottom of the inversion aloft, in m
exp = the exponential function

The above equation includes upward reflection from the ground and downward reflection from the bottom of any inversion lid present in the atmosphere. The sum of the four exponential terms in g_3 converges to a final value quite rapidly. For most cases, the summation of the series with $m = 1$, $m = 2$ and $m = 3$ will provide an adequate solution. σ_z and σ_y are functions of the atmospheric stability class (i.e., a measure of the turbulence in the ambient atmosphere) and of the downwind distance to the receptor. The two most important variables affecting the degree of pollutant emission dispersion obtained are the height of the emission source point and the degree of atmospheric turbulence. The air pollutant concentrations are often expressed as an air pollutant concentration contour map in order to show the spatial variation in contaminant levels over a wide area under study (Beychok, 2005).

2.4.4 Statistical Models

There are a well-defined group of statistical modelling techniques that are available for prediction of algorithm development. The primary techniques can be listed as follows.

(a) Analogue Forecasting (look up table based on past observations) (Carter and Kieslar, 1998)
(b) Regression Analysis (linear, log linear, and stepwise linear) (Slini, 2002)
(c) Cluster Analysis (Abdulmogith, 2003)
(d) Factor Analysis (Killus, 1991)
(e) Discriminant Analysis (Kim, 2000)
(f) K^{th} Nearest Neighbour Analysis (Gooda, 2002)
(g) Classification and Regression Tree Analysis (CART) (Burrow, *et al.*, 1995)

(h) Coupled Models: CART and Regression, Stratified Regression, Regression and Numerical prediction model output statistics
(i) Artificial Neural Networks (ANN) (Gardner, *et al.*, 2001 and Elminir *et al.*, 2006)

The majority of these methodologies rely upon techniques to minimise the variance in data to develop a "best fit" between predictors and the predicated. In this study step-wise multiple linear regression is employed. Regression offers an economical analysis of the data and the resulting product is an easy-to-use model that can be directly applied by a forecaster.

A second component of the modelling methodology that needed resolution was the selection of the predictor variables used for statistical analysis. Inclusion or exclusion of a predictor variable or group of predictor variables to algorithm development can easily alter the path of model generation. In general, many of the meteorological predictor variables are auto zero correlated and compete in the regression based model development. This was very important to stepwise multiple linear regression analysis where the technique incrementally selects the predictor which contributes the greatest to improving the fit of the data. The final model fixes the relationship between the predictor variables and dependent data through regression coefficients. These coefficients apply to the total distribution of data.

2.4.5 Overview of dispersion models used in this research

The candidate models in this research are Industrial Sources Complex Short Term Model (ISCST3) and Atmospheric Dispersion Modelling System (ADMS). They are models that are widely used to evaluate environmental assessment in Thailand and England.

2.4.5.1 The Industrial Sources Complex Short Term model (ISCST3)

Various air quality models are used in Thailand, some are developed by the user themselves to run for a specific purpose, mainly for research in universities. The most widely used air quality model is the Industrial Source Complex - Short Term regulatory air dispersion model (ISCST3), an Eulerian model, developed by the U.S. Environmental Protection Agency (US EPA), which is a Gaussian plume model and is widely used to assess pollution concentration and/or deposition flux on receptors from a wide variety of sources. The ISCST3 model is used for Environmental Impact Assessment (EIA) reports in

where air quality assessment is required. This model used to be recommended by the US EPA. Recently, AERMOD (also developed by the US EPA) is the next generation air dispersion model which incorporates planetary boundary layer concepts. The complex terrain area that consists of mountain, sea, or various types of vegetation can be assessed by a more complex model, the California Mesoscale Puff Model (CALPUFF). This model is a Lagrangian model that handle the wind in 3-dimension so it can treat calm wind. However, upper air data is not monitored in general area in Thailand, so the data can be obtained by simulate meteorological model.

The Gaussian Model

The U.S. EPA has developed computer models since the early 1970s. The basic equation of these models is Gaussian distribution equation:

$$C(x,y,z) = \frac{Q}{2\pi\sigma_y\sigma_z u}\exp\left[-\frac{(y-y_0)^2}{2\sigma_y^2}\right]\exp\left[-\frac{(z-z_0)^2}{2\sigma_z^2}\right] \qquad \text{Equation 2.32}$$

Where:

C (x,y,z) is the downwind concentration at a point x,y,z, g/ m^{-3}

Q is the emission rate of pollutants, g/s

σ_y, σ_z is the plume standard of deviations, m

u is the mean vertical wind speed across the plume height, m/s

y is the lateral distance, m

z is the vertical distance, m

The Industrial Sources Complex Short Term model (ISCST3) is a dispersion model tool for industrial source complex. ISC3 is a steady-state Gaussian plume model which can be used to assess pollutant concentrations from a wide variety of sources associated with an industrial complex. The model, based on the Gaussian equation (Equation 2.32), which is widely used to assess pollution concentration and/or deposition flux on receptors from a wide variety of sources. The plume dispersion configuration is illustrated in Figure 2.23 (ADM, 2004).This model can account for the following: settling and dry deposition of particles; downwash; point, area, line, and volume sources; plume rise as a function of downwind distance; separation of point sources; and limited terrain adjustment. ISC3 operates in both long-term and short-term modes. ISC3 also uses the Emissions Modelling System for Hazardous Pollutants (to process an emission inventory for input into the model). The Building Profile Input Program and the Building Profile Input Program for

PRIME can also be used with ISC3 to correctly calculate building heights and projected building widths for simple, multi-tiered, and groups of structures. (USEPA, 2006)

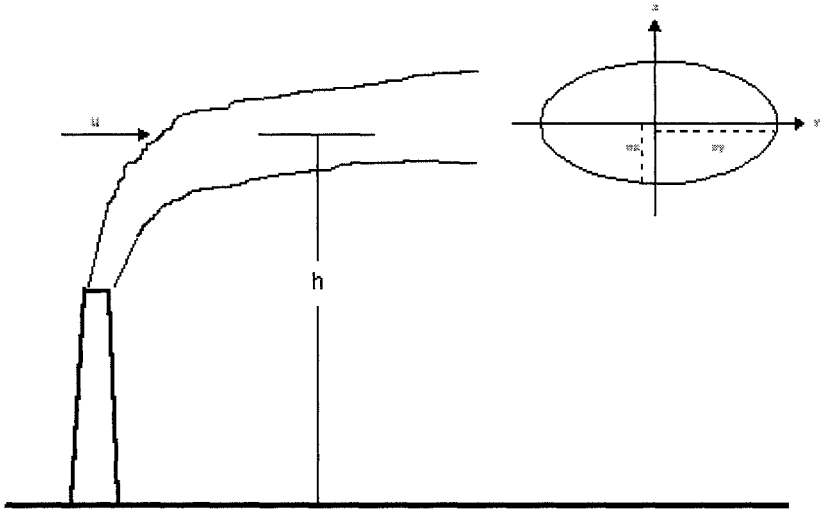

Figure 2.23: Plume dispersion. (EEA, 2003)

Data Requirements

Input data of ISC3 model are emission data, receptor detail, and meteorological data. The emission sources input data of ISC3 are categorised into four basic types of sources, i.e., point sources (stacks, vents, etc.), volume sources (any three dimensional sources, e.g. multiple vent), area sources (circular area, polygon area such as storage pile etc.), and open pit sources (surface coalmine, stone quarry, etc.).

The receptors input data can be gridded and/or discrete receptors in a Cartesian or polar coordinates. Types of grid are uniform Cartesian grid, non-uniform Cartesian grid, uniform polar grid, and non-uniform polar grid.

The required meteorological data in ISC3 are hourly meteorological data: temperature, mixing height, stability class, wind direction, and wind speed.

Factors

The factors that are accounted for in ISC3 are deposition (wet and dry deposition), chemical reaction, building effects (building downwash effect), continuous plume rise model, and terrain.

Limitations

The limitations of ISC3 are

- ISC model cannot calculate concentration at the receptor located within one metre of a point source or an effective radius of a volume source.
- The stack height wind speed (u_s) is not allowed to be 0 m/s.
- The ISC models are not appropriate for estimating concentrations within both highly turbulent and generally recirculation of the air flows in the building cavity region.
- The ISC models print a message and do not calculate concentrations for any source-receptor combination where the source-receptor separation is less than one metre, and also for distances less than three the building height for a squat building, or three projected (crosswind) building width for a tall building under building wake effects.
- The area source algorithm does not provide reliable results for receptors located within or adjacent to very small areas, with dimensions on the order of a few metres across.
- ISCST3 can model within only 50 kilometres.

(U.S.EPA, 1995)

2.4.5.2 Atmospheric Dispersion Modelling System (ADMS)

The ADMS model was developed by Cambridge Environmental Research Consultants (CERC), the UK Meteorological Office, National Power plc and University of Surrey. The first version of ADMS was released in 1993. The current model is ADMS version 4 that was released in 2007. The model can be used for the following regulatory purposes: multiple buoyant or passive industrial emissions, urban or rural areas, flat or complex terrain, transport distances less than 50km, short term (seconds) to annual averaging times.

ADMS 3 is an advanced modified Gaussian dispersion model for calculating ground level concentrations of pollutants emitted both continuously from point, line, volume and area sources, or discretely from point sources that uses a parameterisation of the boundary layer in terms of boundary layer depth and Monin-Obukhov length rather than stability categories. The model includes treatment of building effects, complex terrain, wet and dry deposition, puff (finite duration) releases and short time scale fluctuations in concentration, chemical reactions; radioactive decay and gamma-dose, plume rise as a function of distance, jets and directional releases, averaging time ranging from very short to annual, condensed plume visibility, meteorological pre-processor. A summary of ADMS treatment

is presented in Figure 2.24. Plume rise is calculated by a Runge-Kutta solution of the conservation equations. ADMS uses a Lagrangian plume rise model, a buildings effect model based on a two plume approach, using wake averaged flow values to calculate plume spread (Robins *et al.*, 1997) and a complex terrain module based on the linearised airflow model FLOWSTAR. Dry deposition effects for particles are treated using a resistance formulation in which the deposition velocity is the sum of the resistances to pollutant transfer within the surface layer of the atmosphere, across the laminar sub layer and onto the surface, plus a gravitational settling term. For gases the surface layer resistance is calculated as a function of the nature of the gas: reactive, unreactive or inert. Wet deposition is calculated making use of a specified washout coefficient which may be rainfall rate dependent (USEPA, 2006 and Carruthers, et al., 1994).

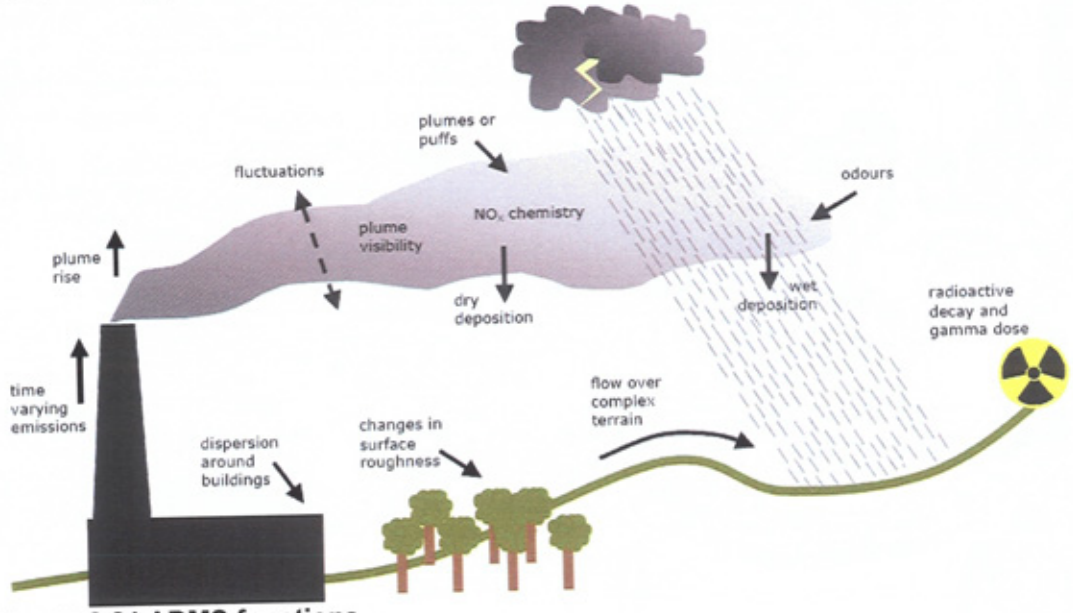

Figure 2.24 ADMS functions.
(source: http://www.cerc.co.uk)

For air dispersion modelling in coastal areas (Figure 2.25), ADMS 3 includes a coastal module that may be invoked when the following conditions are satisfied:

- The sea is colder than the land;
- There are convective meteorological conditions on land;
- There is an onshore wind; and
- The source of interest is above the internal boundary layer calculated at the source location.

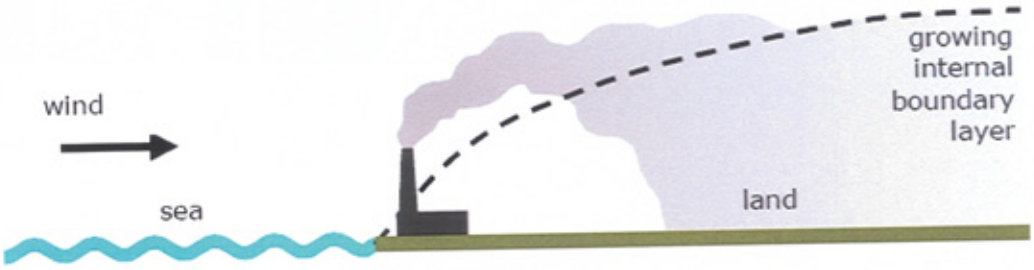

Figure 2.25 Air Dispersion Model in Coastal Area
(source: http://www.cerc.co.uk)

The ADMS family of dispersion modelling programs includes: the original ADMS 3, which is used to model multiple point sources (used in this study); ADMS Roads for modelling traffic emissions; and ADMS Urban which allows the modelling of multiple point and road sources as well as shallow grid sources for large urban conurbations. In addition, several specialist versions have been made available in different parts of the world, for example ADMS-Screen (modelling of a single point source) and ADMS-EIA (multiple point and road sources, developed for Chinese EIA requirements), both of which have been developed for the use in China (McHugh *et al*, 2005). In 2007 a new version of ADMS was released (ADMS 4). New and updated features of the program include: incorporation of a integrated mapper for the visualisation of sources; an increase in the number of sources that can be modelled (from 100 to 300); improved wet deposition model for SO_2; low wind speed modelling capability; improved treatment of meteorological parameters; and a change in the way that NO_x concentrations are expressed (now expressed as 'NO_x as NO_2, which makes the assumption that all of the NO_x is in the form of NO_2).

Numerous validation and evaluatory studies have been carried out on ADMS and its variants. For example, Hanna *et al* (1999) carried out a comprehensive comparative study that looked at the performance of ADMS, AEROMOD and ISC3 in predicting mean and maxima ground level concentrations for five sets of test data representing buoyant and non-buoyant plumes released from sources in rural urban and complex terrain. The test data sets were: OPTEX (non buoyant releases of SF_6 tracer gas from a point source located within a tank farm at an oil refinery, and monitored at points up to 700m downwind); Duke Forest (multiple point sources of SF_6 tracer gas released from an open field, and monitored at points up to 200m downwind); Kincaid (buoyant release of SF_6 tracer gas from a 187m tall stack of a power plant located in flat rural/agricultural terrain,

and monitored at points up to 50m downwind); Indianapolis (buoyant release of SF_6 tracer gas from a 84m tall stack of a power plant located in flat urban/commercial area, and monitored at points up to 12km downwind); and Lovett (buoyant release of SO_2 tracer gas from a 145m tall stack of a power plant located in a valley, and monitored at points up to 3m downwind). For the non-buoyant releases, ISC tends to over-predict the maximum concentrations by up to a factor of ten, whereas ADMS and AEROMOD have broadly similar performances, under predicting the maximum concentration by up to 40%. A useful statistical evaluatory parameter employed is FAC2, which is the fraction of modelled data within a factor of two of the observed values; for the OPTEX set these lay between 0.7 and 0.8 for all three dispersion modelling programs, whereas for Duke Forest, ISC scored 0.17, ADMS 0.63 and AEROMOD 0.53. For the buoyant releases there is mixed performance depending on the site. The FAC2 parameters were as follows. Kincaid: 0.13, 0.59 and 0.29 for ISC, ADMS and AEROMOD respectively; Indianapolis: 0.49, 0.42 and 0.39 for ISC, ADMS and AEROMOD respectively; and Lovett: 0.064, 0.3 and 0.25 for ISC, ADMS and AEROMOD respectively. Overall, ISC was the worst performing of the modelling packages, tending to over predict concentrations; only 20% of predictions were within a factor of two of the observed concentrations. ADMS and AEROMOD were broadly on par with each other, both having a tendency to under-predict concentrations; 53% and 46% respectively of modelled predictions were predicted within a factor of 2. These results should not be surprising, since ISC3 incorporates a much simpler model of vertical turbulence within the atmospheric boundary layer compared with ADMS and AEROMOD both of which use a bimodal distribution of vertical turbulence within the atmosphere.

The satisfactory treatment by ADMS of dispersion of pollutants from a variety of source types in a range of different urban and rural environments has led to its adoption by many regulatory agencies worldwide. ADMS-Urban, which is based on the basic algorithms of ADMS 3, but incorporates additional features such as emissions inventories, increased number of line and point sources, a more complex chemical reaction set, and the ability to model shallow grid sources (e.g. to account for road traffic sources in nearby boroughs without having the complexity of modelling each individual line source). There have been several reported studies in which ADMS Urban has been used to model pollutant concentrations in large urban areas. For example, Owen *et al* (1999) evaluated the performance of ADMS Urban to predict NO_x and SO_2 concentrations in Central and East London; this was part of a project to evaluate ADMS as a tool for local air quality assessment and review, a legal requirement for local authorities in the UK. The modelling area included a 10km radius around the monitoring sites in which significant roads (over

25,000 vehicles per day) were treated as discrete lines sources; but also took into account contributions from areas beyond this radius by aggregating road sources into shallow 1km^2 grids. Results for SO_2, presented as hourly data, showed an over-prediction for the higher percentiles, attributed to the contribution of large power stations in the East Thames corridor, whereas there was reasonable agreement for the mean and 95th percentiles. The percentage of predictions within a factor of two of the observed data ranged from 13.3% to 46%. It was also suggested that the assumption of a steady state in ADMS for models on this scale may not necessarily be valid, i.e., emissions released in one hour might not reach the receptor point within the same hour. For NO_x, there was generally good agreement during the summer (tendency for a slight overestimation), with (FAC2 ranging from 0.43 to 0.59), with poorer results obtained during winter (FAC2 ranging from 0.24 to 0.58), when concentrations were generally under predicted; the authors attributed this trend to the lack of seasonality in the emissions inventory.

The same London study was also reported on by Carruthers *et al* (2000), who compared the results obtained with similar urban dispersion modelling studies carried out at two other locations in the UK – Belfast and Neath/Talbot & Swansea. For the latter study area, which is typical of a heavily industrialised urban environment (oil refining and steel works), it was found that modelling the effects of complex terrain improved the agreement between observed and predicted, particularly for SO_2, though it should be noted that no actual statistical comparison parameters are presented for this pollutant; from the percentile graphs presented it appears that ADMS over-predicts SO_2 concentrations at monitoring stations. Agreement between observed and predicted NO_x concentrations (primarily from traffic sources) are reported to be good, except in low wind speed conditions. Interestingly, the model significantly underestimated the concentrations of PM_{10}, at the monitoring sites; this was attributed to emission sources not being accurately defined, for example not taking into account long range transport of particulate matter, nor fugitive and periodic releases. Benzene was also modelled as part of this study, and was significantly underestimated; the authors again attributed this to problems with defining emission sources (benzene emissions were assumed to be 5% of the total traffic-derived VOC emissions). The final study area covered was Belfast, which represents a lightly industrialised urban environment in which domestic solid fuel consumption contributes significantly to SO_2 and PM_{10} concentrations. The modelling was carried out for a six month winter period where domestic emissions were likely to be significant. Predicted mean concentrations of SO_2 were reported to be in reasonable agreement with measured values, however there was a tendency for the model to over-predict maximum

concentrations (by as much as 100ppb). PM_{10} concentrations were significantly under-predicted, as also observed in the Swansea study, though in this case it was suggested that there was significant conversion of locally emitted SO_2 to sulphate particulates that was not accounted for in the model.

2.5 Summary

This section has reviewed: (a) the ambient air quality monitoring network in Thailand, including the monitoring stations near to the study area; (b) the methodology instrumentation and calibration methods used in the ambient air quality and meteorological stations; (c) current ambient air quality standards in Thailand; (d) the established health and welfare effects of non-methane hydrocarbons, nitrogen oxides, particulate matter, carbon monoxide and sulfur dioxide; (e) time series and panel studies that have established risk factors for short term exposure to air pollutants; (f) cohort studies that have established long term risk factors of exposure to the principal pollutants, especially particulate matter; (g) the use of various statistical approaches to establish correlations between air pollutants and ill health end-points; and (h) dispersion modelling techniques. The latter two reviews have informed our choice of approach for health impact studies for the Maptaphut area and also the choice of atmospheric dispersion model. The next chapter will describe the methodology employed in this study.

Chapter 3

Methodology

3.1 Introduction

This section describes the data collection, analysis and modelling methods used in this study; the overall methodology is summarized in diagrammatic form in Figure 3.1.

There are three main parts to the methodology: data selection (including QA considerations), data analysis, and statistical / dispersion modelling; these are described in detail in subsequent sections.

The methodology is consistent with the achievement of the project objectives, i.e. to characterise meteorological conditions in the Maptaphut area; to investigate the relationship between air pollution and ill health in the Maptaphut area and to evaluate the effectiveness of statistical methods and commercial atmospheric dispersion modelling packages in predicting ground level pollutant concentrations at points surrounding the Maptaphut industrial estate. The latter information can also be used to calculate the contribution of Maptaphut emissions to respiratory ill health in the local community.

All of the objectives have a significant data requirement, and it is important, therefore, that there is confidence in the quality of the data used. The meteorological and ambient air quality data was collected from automated ambient air quality monitoring stations that form part of Thailand's Ambient Air Quality Monitoring Network, as described in Chapter 2; USEPA protocols are followed regarding operation and calibration of the instrumentation, and data validation is carried out by the Thai Pollution Control Department, who collect the data on an hourly basis. The traffic data used is sourced from official Thai Government annual traffic reports. Data on sickness rates was obtained from Maptaphut and Rayong hospitals, albeit without underlying data to indicate sex or age groups, or area of residence. Finally, industrial source emission data used in dispersion modelling studies was obtained from stack monitoring measurements carried by SECOT Ltd on behalf of the Industrial Estate Authority of Thailand; this data is used for regulatory purposes and is carried out in accordance with appropriate US EPA method protocols.

Basic characterisation of meteorology and ambient air quality was carried using Microsoft Excel, MicroMet Plus (Met One Instruments for wind rose generation), and SPSS; Excel was also used to prepare datasets used to characterise the predominant direction of air pollutants at three ambient monitoring stations, and for input into the AirQ Health Impact assessment tool published by the World Health Organisation (WHO European Centre for Environment and Health 2004).

In the final stage of the study, various approaches were explored for the prediction of short term ambient air pollutant concentrations; these involved both statistical models and commercially available dispersion modelling packages. The multiple linear regression function of SPSS was used to develop statistical models that could predict daily maximum ozone and PM_{10} concentrations; this approach was also used to investigate the relationship between ambient air quality and respiratory sickness rates in Maptaphut. The atmospheric dispersion modelling studies evaluated the suitability of three different types of dispersion modelling programs in terms of their ability to predict short term concentrations of sulphur dioxide, nitrogen oxides and PM_{10} at one of the ambient air quality monitoring stations; the evaluation was based on a comparison of predicted concentrations with actual monitored data. The study compared an advanced Gaussian model (ADMS-3), a basic Gaussian model (ISCST3) and a computational fluid dynamics (CFD) model. ISCST3 was chosen as the basic Gaussian model because it has been used for EIA studies in Thailand; ADMS-3 was chosen as the advanced Gaussian model because it has been the subject of several intercomparison studies with ISCST3 (Hanna *et al*, 1999; Hall *et al*, 2000a and 2000b) and because of its availability to the study; PANEIA was chosen as the CFD model, principally because of its availability to the study.

3.2 The Study Area

The Maptaphut Industrial Estates are located in Rayong province, which is about 220 kilometres from Bangkok on Highway No. 3 or 179 Kilometres on Highway No.36 with its new short-cut, and about 80 kilometres east of Pattaya (Figure 3.2). Chonburi is located to its north and west, Chachoengsao to the north, Chantaburi to the east, and the Gulf of Thailand to the south. The population in Rayong is approximately 556,733 people (NSO, 2004). Rayong covers a total area of 3,700 square kilometres with a 100 kilometres long coastline, several beautiful beaches, and scenic waterfalls amid natural surroundings. Rayong province is divided into Muang Rayong, Klaeng, Ban Khai, Bang Chang, Pluak Daeng and Wang Chan districts, and Khao Chamao and Nikhom Phatthana sub-districts.

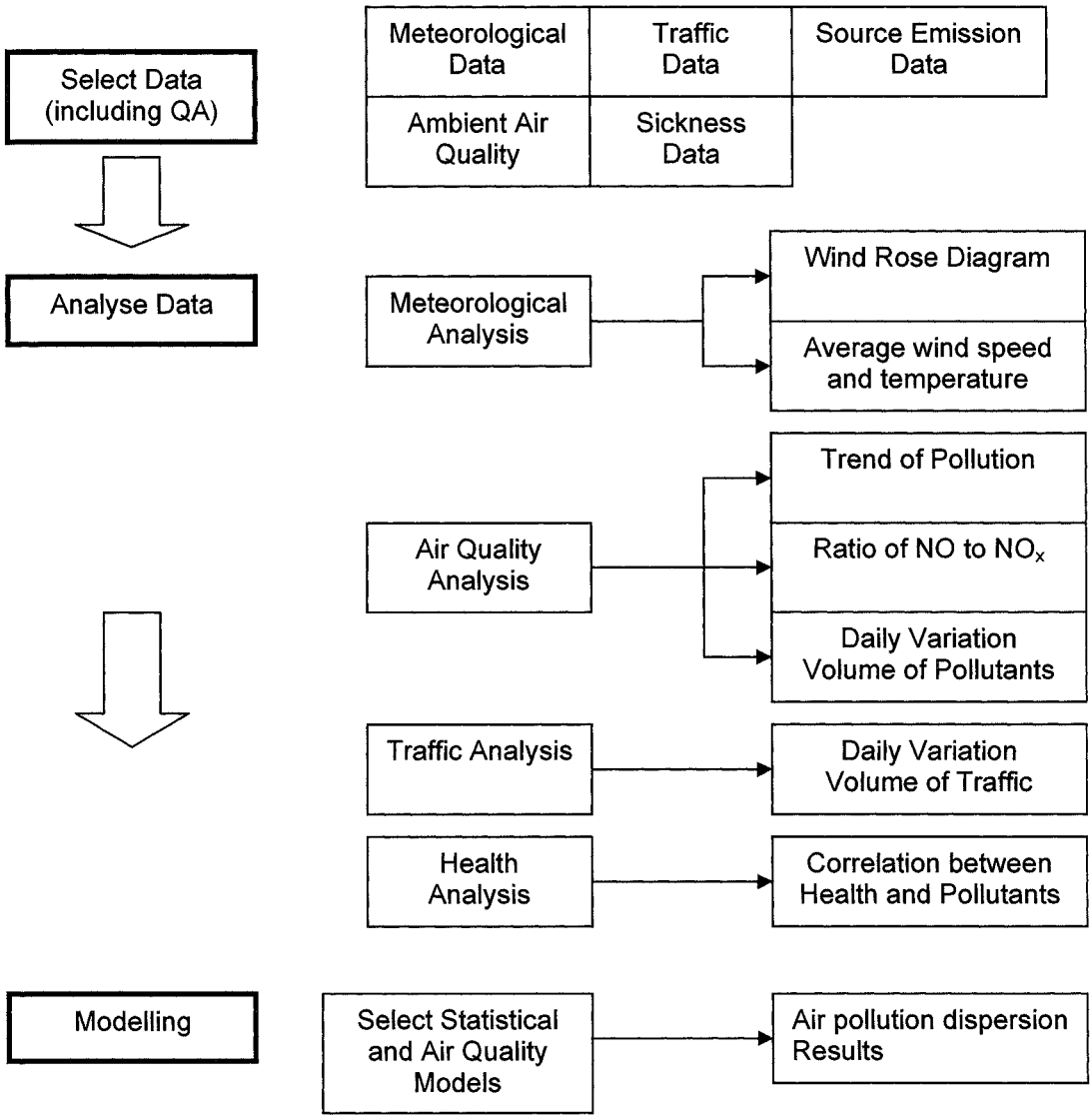

Figure 3.1 Overall Methodology.

Rayong is an important province in the eastern region of Thailand and has been promoted as a major national centre for industry and economic development since 1982, under the Eastern Seaboard Program and the Rayong Provincial Master Plan (MIE, 1999). The area is an ideal location for industrial development due to its good transport links, well developed water resources, and deep seaports. This development plays an important role for the national economy's development in this region, especially in Chonburi and Rayong Province. The rate of industrial expansion in both provinces is extremely high, especially in the centre of the province, namely Sri Racha (Chonburi) and Maptaphut (Rayong).

Large industrial plants such as petrochemical plants, natural gas separation plants, steel production plants, deep sea port, etc. are located in Rayong as a part of Eastern Seaboard Development Program. The total number of industrial plants in Rayong was 1,810 in the end of 2004 (Industrial Works Department, 2005), compared to 1,046 in 1999 (Industrial Works Department, 1999) and 412 in 1988. Among these plants, the large facilities such as petroleum refineries, petrochemical industries, steel production plants, etc. are located in and nearby the Maptaphut Industrial Estates, which is one of the highest priority regional development programs presented since the Fifth National Economic and Social Development Plan. Apart from this industrial estate, Rayong has other industrial plants and agro-industries located in various areas. These developments have resulted in a remarkable change of land use in Rayong Province during the last decade of 20th century.

Figure 3.2 Rayong Province Location on the Map of Thailand. *(Source: The German Thai Link, 2004)*

Remote sensing technology in combination with geographic information systems (GIS) have been used for rendering reliable information of land use change in Rayong during a 10-year period, as presented in Table 3.1. The LANDSAT TM images acquired in 1987, 1990, 1994, and 1997 were classified into six significant land use/land cover types, namely: forest, agriculture, water bodies, industrial area, urban/building, and barren/others. The conversion of forest land to agricultural land and the conversion of agricultural land to water bodies and industrial areas are the main significant changes observed. The water bodies increased from 16.2 km^2 in 1987 to 43.7 km^2 in 1997, while industrial areas extended from 2.1 km^2 in 1987 to 42.8 km^2 in 1997. This picture reflects the national policy and growth of the economy during that time.

Table 3.1 LANDSAT TM satellite data land use in Rayong

Land Use, Land Cover Type	Land areas (square kilometres)			
	1987	1990	1994	1997
Forest	479.3	226.0	327.9	553.3
Agriculture	2,981.5	3,307.3	3,018.6	2,808.0
Water Bodies	16.2	12.8	27.6	43.7
Industries	2.1	15.0	23.6	42.8
Urban / Buildings	63.4	59.6	102.1	86.4
Barren / Others	173.3	95.1	216.0	181.6
Total	3,715.8	3,715.8	3,715.8	3,715.8

Source: Rayong Provincial Office, 1999.

The Maptaphut Industrial Estates occupy a land area of about 9.8 km^2 for about 50 large industries that are mainly the petrochemical industry, chemical and fertilizer industry, steel industry, oil refinery industry, utility industry, and refinery industry (the total area for the estates including residential areas is 16 km^2). Table 3.2 shows a summary of the main industries present, together with the annual production; further details of the industries and their products have been described in Appendix B.

Table 3.2 Industrial Types and their production in the Maptaphut Industrial Estates

Types	Numbers of Plants	Production (tons/year)
Petrochemical Industry	19	5,600,000
Chemical and Fertilizer Industry	15	2,300,000
Steel Industry	7	950,000
Utility Industry	5	900,000
Refinery Industry	2	6,900,000

Source: Maptaphut Industrial Estate, Industrial Estate Authority of Thailand, 1999.

3.3 Data Sources

The main data sources used in the dispersion modelling, health impact analysis and statistical modelling studies carried out as part of this research are summarised in Table 3.3, along with data quality considerations. The data has been obtained mainly from the Thai government, including: the Pollution Control Department, Department of Highway, and Maptaphut Hospital.

3.4 Meteorological analysis

Together with emission sources and rates, large scale and local meteorological conditions are the prime determinants of ambient air quality. Meteorological data was analysed as part of this study using Excel and SPSS. Wind direction data was available from the 100-metre tall meteorological mast and at surface level (10 metres); wind roses were generated using MicroMet Plus (Met One Instruments, USA). Furthermore, average wind speed and temperature were also analysed. In carrying out this analysis one needs to be aware that the industrial estates shown in Figure 3.2, for which modelling studies were carried out, are located in the coastal area; therefore, the role of the sea-land breeze in the dispersion and build up of air pollutants should be considered.

3.5 Status and Trend of Air Pollution in Maptaphut Area

Diurnal air quality status of the Maptaphut and the surrounding area was analysed for a period of ten years from 1998 to 2007. Averaged diurnal variations of hourly pollutant concentrations for each month of the ten consecutive years were calculated and then again averaged to obtain ten year averaged yearly diurnal variation. Averaged diurnal variations of hourly concentrations of NO_x ($NO+NO_2$), O_3, SO_2, PM_{10}, CO, and NMHC at three ambient monitoring stations, 29T, 30T, and 31T station were analysed (the locations of these stations are described in Section 2.1.4. The ratio of NO to NO_x was analysed on the basis of five year averages of the 1998-2002 data. The analysed data was compared with Thai National Ambient Air Quality Standard (NAAQS).

Table 3.3 Data sources and quality considerations

Source	Data	Year	Parameters	Data quality /validity
Pollution Control Department (PCD)	Ambient air quality monitoring data	1998-2006	CO, NO, NO_x, NO_2, SO_2, THC, CH_4, THCNM, O_3, PM_{10} (all ppm with the exception of PM10 in μg m^{-3})	Monitored for regulatory purposes by the Thai Pollution Control Department according to USEPA Code of Federal Regulations 40, part 53. Stations carry out automatic zero and span calibrations of gas analysers every day. Manual calibrations with standard gases complying with USEPA protocol grade carried out every fifteen days. Data collected during periods outside calibration are automatically flagged and are removed from dataset prior to data analysis.
PCD	Meteorological data	1998-2006	WS (ms^{-1}), WD (°), Temp(°C), P(mb), RH (%)	Every six months all meteorological sensors are calibrated as follows: Temperature (Pt-100 sensor) using temperature calibration bath; WS/WD – WS/WD calibrator; RH – RH calibrator using lithium chloride/ sodium chloride; rain gauge by manual calibration; solar sensor - black cover to measure zero. Data collected during periods outside calibration are automatically flagged and were removed from dataset prior to data analysis.
Meteorological Department	Meteorological data	2001-2004	Cloud cover (Octas), mixing height (m), ceiling height (m).	No information, though only used in ADMS runs that required the chemistry module (not critical).
Department of Highway	Traffic data	1997-2006	Average daily traffic, Traffic growth rate	No information.
Maptaphut Hospital and Rayong Hospital	Sickness data	2001-2005	Number outpatients of Respiratory system disease	No data on patient's gender, age or location of residence.
Industrial Estate Authority of Thailand (IEAT)	Source emission data	2004-2007	Unique identifier, UTM coordinates, stack height, stack diameter, temperature (°C), exit velocity (m/s), emission rates (g/s) for SO_2, NO_x and PM_{10}.	Stack monitoring carried out for regulatory purposes by SECOT Ltd on behalf of IEAT. All stack monitoring carried out according to US. EPA Method 5: Determination of Particulate Emissions, Method 6: Sulphur Dioxide emissions determination and Method 7: Nitrogen Oxide emission determination.

Note: CO = Carbon monoxide, NO = Nitrogen monoxide, NO_x = Nitrogen oxides, NO_2 = Nitrogen dioxide, SO_2 = Sulphur dioxide, THC = Total Hydrocarbon, CH_4 = Methane, THCNM = Total non-methane hydrocarbons, O_3 = Ozone, PM_{10} = Particulate matter less than ten microns, WS = Wind speed, WD = Wind direction, Temp = Temperature, P = Pressure, RH = Relative Humidity.

3.6 Diurnal Status of Traffic

The average daily traffic (ADT) data (Table 3.4) was obtained from the Department of Highway for Route Number 3 and 36, both of which pass through the Maptaphut Area (Figure 3.3). As was seen from Figure 2.1 in Chapter 2, Air Quality Monitoring Stations are located near to Route 3.

The Pollution Control Department classified the roads in Thailand into seven area types and each type is sub-divided into four sub-types depending on the number of lanes. A total of 30 road types are classified in this way. Area type 2 (Medium population density and urban area) was selected for Route 3 and Area type 3 (Low population density and urban area) was selected for Route 36. The number of lanes is also taken into account (four for both routes), giving road type numbers of six and ten for Route 3 and Route 36, respectively. The expansion factor (F) for these road types is shown in Table 3.5 (PCD, 1994).

The following relation between ADT and expansion factor (PCD, 1994) is used to find the hourly volume (HV) of traffic.

$$ADT = 15 \times F \times HV \times NHF \qquad \text{Equation 3.1}$$

Where

ADT = average daily traffic volume PCU/day

F = expansion factor for different hour (06:00 to 21:00)

HV = hourly volume PCU/hr

NHF = night hour expansion factor

For time of 21:00 to 6:00, the equation is:

$$ADT = 15 \times F \times HV \times NHF / 9 \qquad \text{Equation 3.2}$$

Where

F = expansion factor for 21:00 to 6:00

HV = average hourly volume for 9 hour period (21:00 to 6:00)

The PCU unit used is the Passenger Car Equivalent Unit and Table 3.6 shows the factors used to calculate the equivalent PCU for various types of vehicles. By using this factor and neglecting the bicycle and tricycle (since the amount is very small and their emission factor is zero), the ADT in PCU unit is found, as listed in Table 3.7.

Then using equations 3.1 and 3.2, ADT data, and expansion factor for classified routes, hourly traffic volume in PCU is obtained.

3.7 Health Impact Relationships for Air Pollution

Causal links between air pollution and health effects are difficult to establish due to the many factors that are involved such as: the type of pollutants, length of exposure, extent of interaction among pollutants, as well as other environmental and non-environmental factors. Specially designed studies on exposure assessment are normally required for the purpose; many such studies have been reviewed in Chapter 2. The limited health data available in this study was not sufficient to draw firm conclusions; however it did permit a simple correlation study to be carried between air pollution levels in Rayong / Maptaphut and weekly annual sickness rates.

Air pollution is a possible cause of sickness in the studied area. The sickness data of the Maptaphut Hospital, which is located in the Maptaphut Industrial Estate area, is shown in Table 3.8. The sickness data included outpatients of the Rayong Hospital, located in Rayong downtown. The studied period is three years (2003-2005) The top ten sicknesses of the local people are: respiratory system disease, digestive system disease and oral cavity disease, skin disease and subcutaneous, infectious diseases and parasites, muscle structure disease, circulation system disease, endocrine system disease, eye disease, genital and urinary disease, and psychosis and behavioural disease. Of all these diseases, the people in this area had the most serious problem with their respiratory system. The relationship between respiratory disease and air pollutant levels was primarily studied by the Multiple Regression correlations method.

Multiple Regression correlation analysis was used for the purpose of predicting the relationship between respiratory outpatient admission and air pollutants as well as temperature and %RH. Exposure time to pollutants was also considered, so separate correlations were carried out for different averaging times, for example one week and two weeks. Multiple Linear Regression was carried out using SPSS statistical software.


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อำเภอสัตหีบ
จังหวัดชลบุรี
จังหวัดระยอง
อำเภอเมืองระยอง
อำเภอบ้านค่าย
ปณ. สัตหีบ
ปณ. อู่ตะเภา
ตลาดสัตหีบ
หาดสัตหีบ
หาดนำริน
หาดพัทยา
เกาะสัตหีบ
เกาะขาม
เกาะแรด
เกาะแสมสาร
วิทยาลัยอาจนิลสัตหีบ
สถานีรถไฟชุมทางเขาชีจรรย์
สถานีรถไฟบ้านฉางหลวง
ตลาดสายลม
ตลาดมาบตาพุด
ตลาดบ้านค่าย
ตลาดเนินพระ
ตลาดเชิงเนิน
ตลาดกะเฉด
ตลาดปากน้ำกระแส
ตลาดวังจันทร์
ตลาดเขาชะเมา
ตลาดเขาน้อย
ตลาดเขาไม้แก้ว
ตลาดเขาหินซ้อน
ตลาดเขาหินลาด
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Table 3.4 Average Daily Traffic (ADT) and Traffic Growth Rate (GR) on Route 3 and Route 36 for road sections near Maptaphut 1997-2001

NO	ROUTE Control	NAME	STATION (m+km)	YEAR	CAR TAXI	LIGHT BUS	HEAVY BUS	LIGHT TRUCK	MEDIUM TRUCK	HEAVY TRUCK	TOTAL	HEAVY VEH %	TRIC+ BICYCLE	MOTORCYCLE
1	3	Km. 186+000 (CHONBURI DIST) – JCT TO BAN KHAI ROAD-SUKHUMWIT TRAFFIC 4 LANES	206+000	1997	17,711	2,889	2,470	21,752	4,615	3,263	52,700	19.64	13	8,939
				1998	20,380	4,226	3,864	22,343	4,420	2,959	58,192	19.32	9	12,013
				1999	22,553	5,455	4,708	22,789	4,801	3,896	64,202	20.88	13	26,489
				2000	19,233	5,185	2,976	14,428	2,317	1,591	45,730	15.05	217	11,830
				2001	13,214	430	669	9,501	893	3,668	28,375	18.43	56	4,087
2	36	KM22+705 (CHONBURI2 DIST) – BYPASS RAYONG TRAFFIC 4 LANES	38+200	1997	12,492	3985	4073	6449	4415	4023	35437	35.30	14	4219
				1998	9492	823	661	681	1086	1321	14064	21.81	21	906
				1999	13493	1127	689	649	1476	1225	18659	18.17	7	975
				2000	7182	974	265	8692	2782	5985	25880	34.90	55	612
				2001	6598	1022	288	8994	2334	6211	25447	34.71	47	676

Source: Department of Highway

Table 3.5 Expansion Factor (F) for Route 3 and Route 36

Time	Route 3	Route 36
6-7	2.3853	2.265
7-8	0.6523	0.6894
8-9	0.7384	0.7836
9-10	0.9686	0.8873
10-11	1.0184	0.9172
11-12	1.0263	0.9629
12-13	1.1492	1.0687
13-14	1.0767	1.0275
14-15	0.9521	0.958
15-16	0.8971	0.9079
16-17	0.7837	0.7852
17-18	0.7495	0.8584
18-19	1.0872	1.0516
19-20	1.4398	1.5715
20-21	1.9371	2.0161
21-6	0.8208	0.7427
NHF	1.0812	1.0898

Source: PCD, 1994.

Table 3.6 Passenger Equivalent Factor by vehicle type

No.	Vehicle Type	PCU
1	Bicycle	0.33
2	Tricycle	0.50
3	Motorcycle	0.33
4	Minicar (Samlor, Silor)	1.00
5	Car	1.00
6	Light Truck (Pickup)	1.00
7	Minibus	1.25
8	Bus	2.5
9	Truck	2.5

Table 3.7 Average Daily Traffic in Passenger Car Equivalent Unit /day for Route 3 and Route 36

Year	Route 3	Route 36
1997	71,883	56,592
1998	80,077	19,170
1999	93,914	24,348
2000	61,256	39,876
2001	37,676	39,175
Average	**68,961**	**35,832**

Table 3.8 Respiratory disease admission data of outpatients at Rayong Hospital

	Jan				Feb				Mar			
	week 1	week 2	week 3	week 4	week 1	week 2	week 3	week 4	week 1	week 2	week 3	week 4
Year	1 to 7	8 to 14	15 to 21	22 to 31	1 to 7	8 to 14	15 to 21	22 to 28	1 to 7	8 to 14	15 to 21	22 to 31
2003	879	991	1041	1330	925	878	742	721	587	795	597	1045
2004	1023	1137	1167	1876	1137	1111	1115	1019*	792	893	738	1008
2005	924	928	878	1275	864	856	931	805	906	848	757	1152
	Apr				May				Jun			
	week 1	week 2	week 3	week 4	week 1	week 2	week 3	week 4	week 1	week 2	week 3	week 4
Year	1 to 7	8 to 14	15 to 21	22 to 31	1 to 7	8 to 14	15 to 21	22 to 28	1 to 7	8 to 14	15 to 21	22 to 31
2003	808	672	568	834	631	635	669	1235	1062	1030	1044	1667
2004	576	404	554	705	534	766	734	1095	798	916	1086	1434
2005	701	592	653	841	649	740	729	1352	1013	1041	1015	1239
	Jul				Aug				Sep			
Year	week 1	week 2	week 3	week 4	week 1	week 2	week 3	week 4	week 1	week 2	week 3	week 4
	1 to 7	8 to 14	15 to 21	22 to 31	1 to 7	8 to 14	15 to 21	22 to 28	1 to 7	8 to 14	15 to 21	22 to 31
2003	1506	1434	1541	1911	1401	1183	1115	1415	1077	940	862	1303
2004	1000	969	964	1376	828	856	860	1286	956	978	1147	1656
2005	976	978	818	1031	826	785	776	1145	770	831	881	1026
	Oct				Nov				Dec			
Year	week 1	week 2	week 3	week 4	week 1	week 2	week 3	week 4	week 1	week 2	week 3	week 4
	1 to 7	8 to 14	15 to 21	22 to 31	1 to 7	8 to 14	15 to 21	22 to 28	1 to 7	8 to 14	15 to 21	22 to 31
2003	851	862	960	1115	711	805	876	963	826	842	922	1425
2004	1300	1192	1054	1089	797	770	797	1071	770	834	828	1210
2005	-	-	-	-	-	-	-	-	-	-	-	-

* date 22-29 February 2004
Source: Rayong Hospital, 2003-2005

Table 3.9 Respiratory disease admission data of outpatients at Maptaphut Hospital

	Jan				Feb				Mar			
	week 1	week 2	week 3	week 4	week 1	week 2	week 3	week 4	week 1	week 2	week 3	week 4
Year	1 to 7	8 to 14	15 to 21	22 to 31	1 to 7	8 to 14	15 to 21	22 to 28	1 to 7	8 to 14	15 to 21	22 to 31
2003	191	269	299	361	274	283	167	195	144	167	210	304
2004	235	238	270	441	294	254	295	213*	155	202	205	197
2005	336	373	344	491	330	363	385	360	400	355	369	603
	Apr				**May**				**Jun**			
	week 1	week 2	week 3	week 4	week 1	week 2	week 3	week 4	week 1	week 2	week 3	week 4
Year	1 to 7	8 to 14	15 to 21	22 to 30	1 to 7	8 to 14	15 to 21	22 to 31	1 to 7	8 to 14	15 to 21	22 to 30
2003	246	180	150	230	197	194	241	415	277	275	312	403
2004	137	100	117	144	151	226	232	359	275	282	316	417
2005	347	296	329	390	298	357	384	738	431	444	434	561
	Jul				**Aug**				**Sep**			
	week 1	week 2	week 3	week 4	week 1	week 2	week 3	week 4	week 1	week 2	week 3	week 4
Year	1 to 7	8 to 14	15 to 21	22 to 31	1 to 7	8 to 14	15 to 21	22 to 31	1 to 7	8 to 14	15 to 21	22 to 30
2003	395	452	495	514	374	283	271	364	241	282	262	310
2004	287	283	327	427	279	263	288	414	305	368	348	453
2005	435	460	429	523	418	350	334	462	419	393	389	556
	Oct				**Nov**				**Dec**			
	week 1	week 2	week 3	week 4	week 1	week 2	week 3	week 4	week 1	week 2	week 3	week 4
Year	1 to 7	8 to 14	15 to 21	22 to 31	1 to 7	8 to 14	15 to 21	22 to 30	1 to 7	8 to 14	15 to 21	22 to 31
2003	333	296	328	291	219	203	216	265	210	196	225	320
2004	420	367	401	433	248	226	255	374	297	272	330	408
2005	434	501	449	506	385	394	493	639	412	426	406	530

* date 22-29 February 2004
Source: Maptaphut Hospital, 2003-2005

3.8 Statistical Model for PM_{10} and O_3

In addition to the dispersion modelling studies described in the following section, multiple linear regression models were also developed to predict local concentrations of PM_{10} and O_3, which are the two pollutants associated with the greatest health effects; the maximum concentrations of both of these pollutants were found to exceed the National Ambient Air Quality Standards (NAAQS) in Rayong province during the dry season. A stepwise linear regression analysis was performed on the data at all three ambient air quality monitoring stations in Rayong province to predict the daily maximum 1-h O_3 and PM_{10}. Meteorological data included global radiation (GR), relative humidity (RH), surface temperature, wind speed at 10, 50 and 100m (WS10, WS50, and WS100, respectively) recorded at the 100m tall meteorological mast. Only data for dry season in the period from1998 to 2002 were used.

The stepwise regression procedure was utilised because it identified the optimum subset of the independent variables to be incorporated into the regression equation. At each step of the variable selection procedure, the variable was chosen when it had the largest partial correlation coefficient with the daily maximum 1-h O_3 and PM_{10} concentration (after allowing for those variables previously selected). The *F* statistic for each of the independent variables to be entered into the model was calculated and its *p* value was then compared to the entry significance level of 0.05. The variables that were significant at the entry significance level would then be allowed enter into the model. The stepwise procedure would examine all previously entered variables and remove any variables whose *F* statistic was not significant at the retention significance level of 0.01. The procedure terminated when no remaining variables had the *F* statistic significant at the entry level and every variable in the model was significant at the retention level.

The accuracy of the statistical model in predicting future concentrations, and in predicting concentrations in other geographical areas was determined by time series analysis. Observed data and predictions from the statistical data were compared yearly during 1998-2002 (based year of statistical model) at the ambient air quality monitoring stations 29T, 30T and 31T located nearby the Maptaphut Industrial Estate, Rayong province. Comparisons of time series for observed against predicted maximum PM_{10} and O_3 concentrations were made for Rayong province (station 29T, 30T and 31T) in 2003 and 2004 and at Chonburi province (station 32T and 33T) in 2004. Chonburi

Province is located next to Rayong in the western part; around 100 km from Bangkok (see Figure 3.2).

3.9 Dispersion modelling

The principal objective of the dispersion modelling studies outlined in this section was to evaluate the effectiveness of a range of different modelling packages in predicting ground level pollutant concentrations at points surrounding the Maptaphut industrial estate. This was to be done by comparing predicted pollutant concentrations (NO_x, SO_2 and PM_{10}) with those measured at the 29T ambient air quality monitoring station located in a residential area adjacent to the Maptaphut Industrial Estate. The results of the evaluation can then be used to advise the appropriate authorities on the suitability of different modelling packages for (a) the prediction of short and long term ambient ground level pollutant concentrations, including whether currently used models are adequate and (b) the estimation of the overall contribution of atmospheric pollutants from large industrial estates towards respiratory illness rates in neighbouring residential areas (in conjunction with the WHO health impact assessment tool, AirQ). Dispersion models are important tools in the prediction of ambient pollution levels for regulatory purposes, however different models can predict markedly different ambient levels given the same source and meteorological data (Hall *et al*, 2000a). Many intercomparison evaluation studies have been carried out on different dispersion models, however these are often based on test sets gathered for only one elevated stack in flat terrain and for limited meteorological conditions (see for example the Kincaid test set evaluations reported on by Hanna *et al*, 1999). The present study attempts an intercomparison for a large industrial estate comprising over 200 individual stacks under a range of meteorological conditions including winter and summer monsoon conditions. In this respect it should be considered a much more realistic intercomparison study, and one specific to South East Asian climatic conditions.

Three dispersion models were selected for use in this intercomparison study: ADMS 3 (CERC, UK) , ISCST3 (US EPA, United States) and PANEIA (Fluidyn, France). ISCST3 was used as a baseline model since it has been widely used in Thailand and South East Asia (e.g. Carrillo and Peralta, 2003), particularly for Environmental Impact Assessments. The model, as discussed in Section 2.4.5, is an older Gaussian type model that incorporates a basic treatment of boundary layer stability state based on Pasquil/Gifford categories; the stability is calculated from the raw meteorological conditions (Hall *et al*, 2000a and 2000b). ISCST3 has been the subject of several intercomparison studies, particularly with ADMS and AEROMOD, both often

categorised 'advanced Gaussian models' because of their similar treatment of atmospheric stability in terms of the Monin-Obukhov length scale to take into account atmospheric stratification effects (Hall *et al* 2000a and 2000b; Hanna *et al*, 1999). In this context, the selection of an advanced model against which to compare ISCST3 would logically be from either ADMS or AEROMOD (which is a USEPA replacement for ISCST3); ADMS was selected because of it's availability to the study and because of the researchers' familiarity with it. Future work could involve the use of AEROMOD to complement the present evaluation studies. Finally it was decided to include a non-Gaussian model in the intercomparison study, specifically a computational fluid dynamics (CFD) model, so that a completely different approach could be compared. The study had access to a short-term licence for Fluidyn PanEIA and so this was the model chosen; it has been the subject of some intercomparisons with ADMS, ensuring some literature data with which to compare the evaluations (based on the Kincaid dataset: Fluidyn, 2006).

Full logs were kept detailing the mode, run number, meteorological data set, model parameters and settings used in the modelling runs. Example logs are shown in Tables 3.11, 3.16 and 3.19.

3.9.1 Dispersion modelling using ADMS 3

ADMS 3 is one of a range of new generation steady state Gaussian modelling programs that is applicable to the modelling of multiple point sources. Related programs include ADMS Roads for air quality predictions around road sources and ADMS-Urban which has the capacity to model point and road sources in a large urban catchment area (see Owen *et al* (1999) for an example of the model being applied to the Greater London region).

A fairly simple model was used to model air quality around the Maptaphut industrial estate. The study area comprised fairly flat terrain and insufficient topographical data was available; therefore, the 'hills' module of ADMS was not utilised. Some limited sea temperature data was available from the Royal Thailand Navy, though the depth, at which the measurements were taken, 3 metres. In any case, ADMS 3 'crashed' when a sea temperature input file was input for use in the 'Coastline' module that simulates the sea-air breeze. The 'Chemistry' module was utilised for some runs, but was found to be unsatisfactory in modelling either ozone or nitrogen dioxide concentrations (very low levels of both pollutants were generated by the model, and bore no resemblance to the actual measured concentrations).

Hourly sequential meteorological data was obtained from monitoring station 29T (see Figure 3.4 for location). It was converted into a file type suitable for ADMS, as shown in Table 3.10, and included the year (YEAR), Julian day number (TDAY), the hour of the day (THOUR), wind speed (U in ms^{-1}), wind direction (PHI in degrees), temperature (TOC in °C), relative humidity (RHUM in %) and the surface sensible heat flux (FTHETA0 in Wm^{-2}). In addition, a separate file was constructed for those cases where chemistry was to be modelled; this module required cloud cover (CL in Oktas) data to be entered. One disadvantage of ADMS compared to computational fluid dynamics programs is the inability of the former to calculate solutions at low wind speeds (less than 0.75m/s).

Figure 3.4 Location of 29T ambient air quality monitoring station.
Source: map from Mapmagic Thailand

Table 3.10 Example of part of an ADMS-compliant meteorological data file

VARIABLES:
9
YEAR
TDAY
THOUR
U
PHI
T0C
RHUM
FTHETA0
P
DATA:
2004,1,0,1.12,298,25,86,-29,0
2004,1,1,0.88,297,24.7,87,-29,0
2004,1,2,0.89,304,24.5,88,-29,0
2004,1,3,1.05,297,24.3,88,-29,0
2004,1,4,1.02,297,24,88,-29,0
2004,1,5,1.21,308,23.9,86,-29,0
2004,1,6,1.01,296,23.6,86,-26,0
2004,1,7,1.15,302,24.6,82,32,0
2004,1,8,1.65,335,27.4,69,114,0

Emission data was obtained from the Industrial Estate Authority of Thailand. Data was available for emissions of SO_2, NO_x and PM_{10} for over two hundred separate point sources in the Maptaphut industrial estate. In addition to pollutant emissions (g/s), the UTM coordinates, height, internal diameter, emission temperature and emission velocity of each stack was detailed. Unfortunately ADMS 3 can handle only 100 point sources and so the 100 most significant sources were selected; the selected sources represented at least 95% of each of the three pollutant types. Emission data was manually entered. Other important modelling parameters were as detailed in Table 3.11. Short term and long term modelled data was generated for the years 2002 to 2005. Where long term values were required for comparison to Ambient Air Quality Standards, these were as specified in Table 3.12

In addition to the ADMS 3 modelling work, the contribution of the main highway routes 3 and 36 to the nitrogen dioxide levels at monitoring point T29 was calculated using ADMS-Roads and the time varying traffic emission data specified in Table 3.4 and 3.5. Parameters were the same as used in the ADMS 3 modelling work (see Table 3.11). The calculated hourly values for road traffic sources were added to the corresponding values calculated for the point sources using ADMS. Other variations were to use the chemistry module to calculate ozone concentrations and to vary the surface roughness.

Calculated concentrations at the monitoring points were compared with actual monitoring data collected at these points. Comparisons were made at the level of:

hourly data, daily averages and 5-day rolling daily averages. The latter comparison allows the data to be smoothed out so that the general comparison of trends over the course of a year can be viewed more easily. Long term concentrations were visualised in the form of contour plots by linking ADMS to ArcView GIS 3.2; in this way the likely health effects on the residential areas neighbouring the industrial estate could be easily determined.

Table 3.11 ADMS modelling parameters

Parameter	Value
Surface roughness at source and monitoring station	1.5m (large urban areas)
Latitude	12°
Dry deposition	No
Wet deposition	No
Chemistry	No
Buildings	No
Hills	No
Coastline	No
Time varying emissions	No
Location of meteorological data collection	29 T monitoring station (UTM 735500, 1405500)
Height or recorded wind	10m
Minimum Monin Obukhov length	30m (cities and large towns)
Surface albedo	0.23 (model default)
Priestley-Taylor parameter	1 (model default)
Output grid x	726954 to 756248, 50 points
Output grid y	1397740 to 1416878, 50 points
Output z value	2m

Table 3.12 Long term values generated by ADMS for comparison with Thai Ambient Air Quality Standards:

Pollutant	Standard	Comparison value
NO_2	1hr average (yearly maximum, specified as 100th percentile in ADMS)	170 ppb (320 μg m^{-3})
SO_2	Yearly mean	40 ppb 100 μg m^{-3})
	24hr average (yearly maximum, specified as 100th percentile in ADMS)	120 ppb.(300 μg m^{-3})
	1hr average (yearly maximum, specified as 100th percentile in ADMS)	300 ppb.(780 μg m^{-3})
PM_{10}	24hr average (yearly maximum, specified as 100th percentile in ADMS)	120 μg m^{-3}
	Yearly mean	50 μg m^{-3}

3.9.2 Dispersion modelling using ISC

Industrial Source Complex model, ISC, is an alternative model recommended by USEPA. The ISC model has been used in regulatory applications in Thailand, for example in EIA modelling for planning applications submitted to the Office of Natural Resources and Environmental Policy and Planning (ONEP).

There are two basic input data requirements, namely the input 'runstream' file and the meteorological data file. The 'runstream' setup file contains the selected modelling options, as well as source location and parameter data, receptor locations, meteorological data file specifications, and output options (USEPA, 1995).

The ISC model was run to investigate air pollutant dispersion from the Maptaphut industrial estate and to compare this with ADMS results; the model domain and data set of ADMS and ISC models were the same.

The input meteorological data source was obtained from the same surface station as ADMS (29T station see Figure 3.4 for location) belonging to the Pollution Control Department. The input data are hourly wind speed (m/s), wind direction (degree), temperature (degree Kelvin). The upper air data was obtained from the Meteorological Department, which included cloud cover (tenth) and mixing height (feet). The mixing height data is essential for ISC modelling: for times where this data is missing, no result can be generated. In contrast, ADMS is able to predict mixing height from other meteorological data. The mixing height data was available until year 2004 only. All data was rearranged format to input ISC by PCRAMMET program (see Figure 3.5). Wet and dry deposition was not considered in the running of the model.

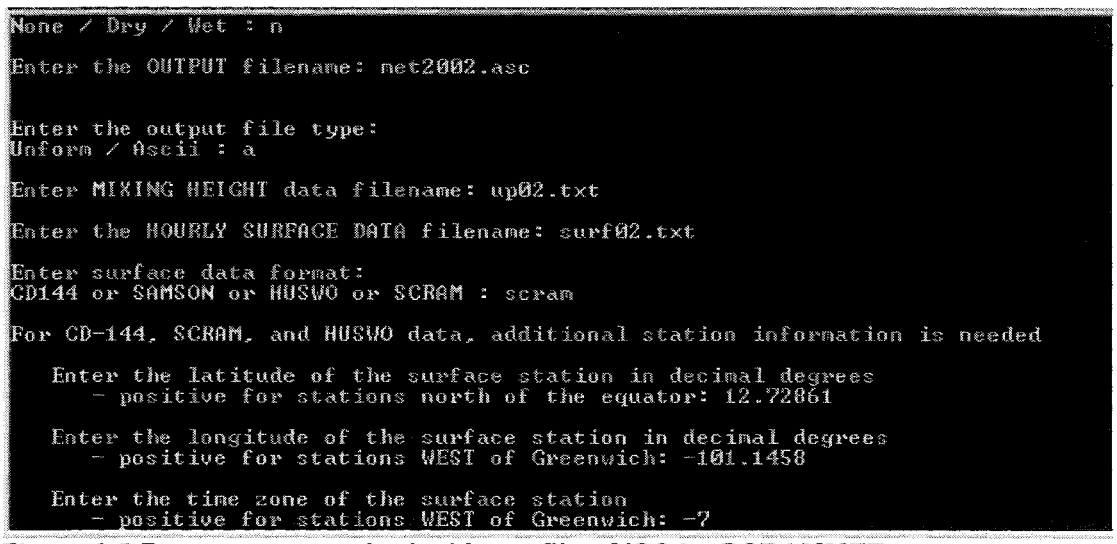


Figure 3.5 Prepare meteorological input file of ISC by PCRAMMET.

The input files of PCRAMMET model comprises the surface and upper air meteorological data; example input files for surface and upper air meteorological data is shown in Tables 3.13 and 3.14. An example from PCRAMMET is presented in Table 3.15. In Table 3.13 the surface meteorological data contains the station code, year, month, date, hour, wind direction (tens degree), wind speed (knots), temperature (F) and cloud cover. Upper air data in Table 3.14 comprises the station code, year, month, date, mixing height value in the morning (feet at 7.00 am) and mixing height value in the afternoon (feet at 13.00 pm). The output file of PCRAMMET consists of year, month, date, hour, wind direction (degree), wind speed (m/s), temperature (K), stability class (0-7), and rural and urban mixing height (feet).

Table 3.13 Example of surface meteorological input data of PCRAMMET

```
1111102010100000330000660101
1111102010101025000000660101
1111102010102025330010650101
1111102010103025330000650101
1111102010104021340000640000
1111102010105021340000640000
1111102010106021360000640000
1111102010107025340000640202
1111102010108025350010670202
1111102010109025360010740202
1111102010110025010010790303
1111102010111025030010830303
1111102010112025030020850303
1111102010113025040020870303
1111102010114025140030860303
1111102010115025140030840303
1111102010116025170020830202
1111102010117025210020820202
1111102010118025240010790202
1111102010119000260010770101
1111102010120000300000740101
1111102010121000040000710101
1111102010122000340000700000
1111102010123000340000690000
1111102010200000340000 6900
```

Table 3.14 Example of upper air meteorological input data of PCRAMMET

```
13100011231  0600            1500
13100020101  0500            1500
13100020102  0700            1550
13100020103  0700            1600
13100020104
13100020105  0450            1550
13100020106  1100            1850
13100020107  0600            1550
13100020108  0450            1750
13100020109  0800            1600
13100020110  0800            1650
13100020111  1050            1800
```

```
13100020112  0750              1650
13100020113
13100020114  0700              1650
13100020115  0550              1550
13100020116  1150              2450
13100020117  0850              1150
13100020118  0600              1750
13100020119  0450              0800
13100020120  0600              1550
13100020121  0800              1350
13100020122  0700              1450
13100020123  1000              1700
13100020124  0800              1750
13100020125  0450              1750
13100020126  0650              1350
```

Table 3.15 Example of PCRAMMET output

```
11111   2002  13100   2002
02 1 1 1 151.0000   0.0000 292.0 7 1500.0  500.0
02 1 1 2 148.0000   1.0000 291.5 7 1500.0  500.0
02 1 1 3 154.0000   0.0000 291.5 7 1500.0  500.0
02 1 1 4 163.0000   0.0000 290.9 7 1500.0  500.0
02 1 1 5 163.0000   0.0000 290.9 7 1500.0  500.0
02 1 1 6 182.0000   0.0000 290.9 7 1500.0  500.0
02 1 1 7 165.0000   0.0000 290.9 6   65.3  543.5
02 1 1 8 173.0000   1.0000 292.6 5  270.2  680.1
02 1 1 9 177.0000   1.0000 296.5 4  475.2  816.8
02 1 110 191.0000   1.0000 299.3 3  680.1  953.4
02 1 111 214.0000   1.0000 301.5 2  885.1 1090.1
02 1 112 206.0000   1.0289 302.6 2 1090.1 1226.7
02 1 113 223.0000   1.0289 303.7 2 1295.0 1363.4
02 1 114 319.0000   1.5433 303.2 2 1500.0 1500.0
02 1 115 322.0000   1.5433 302.0 2 1500.0 1500.0
02 1 116 354.0000   1.0289 301.5 2 1500.0 1500.0
02 1 117  31.0000   1.0289 300.9 3 1500.0 1500.0
02 1 118  57.0000   1.0000 299.3 4 1500.1 1500.1
02 1 119  84.0000   1.0000 298.2 5 1502.6 1360.6
02 1 120 117.0000   0.0000 296.5 6 1505.1 1228.5
02 1 121 220.0000   0.0000 294.8 7 1507.6 1096.4
02 1 122 162.0000   0.0000 294.3 7 1510.1  964.3
02 1 123 160.0000   0.0000 293.7 7 1512.6  832.1
02 1 124 160.0000   0.0000 293.7 7 1515.1  700.0
```

Emission data was the same source as used in the ADMS model. The emission data was input manually and included: the coordinate of stack, elevation (metres), stack height (metres), temperature (K), diameter (metres), exit velocity (m/s), and emission rate of each pollutant (g/s). Other parameters for the ISC model are presented in Table 3.16. Modelling was carried out over both monthly and yearly time periods for the years 2002 to 2004.

Table 3.16 ISC modelling parameters

Parameter	Value
Surface roughness at source and monitoring station	yes
Dry deposition	No
Wet deposition	No
Chemistry	No
Buildings	No
Hills	No
Coastline	No
Time varying emissions	No
Location of meteorological data collection	29 T monitoring station (UTM 735500, 1405500) and Meteorological Department (latitude12°, longitude 101°)
Height or recorded wind	10m
Output grid x	50 grids
Output grid y	50 grids
Output z value	No

3.9.3 Dispersion modelling using PANEIA

The final set of dispersion modelling studies was carried out using a commercially available computational fluid dynamics dispersion modelling package, PANEIA; it is marketed specifically for modelling the air pollution aspects of environmental impact assessments. Like ADMS 3, the use of line sources is limited and so the effect of traffic pollution sources cannot be evaluated (an alternative program called PANAIR can be used for this).

Emission data was converted from an Excel file to an ASCII file in suitable format as shown in Table 3.13 for the first three of the sources. It should be noted that in PANEIA the UTM coordinate system is not used and so all UTM coordinates were converted to coordinates on a 8400 x 11100m grid. A further difference to ADMS is that pollutant concentrations are expressed as μg/g of air emitted from the stack, rather than as a mass flow rate (g/s); a conversion was carried out on the emission data file. Meteorological data from monitoring station T29 was also converted to a compatible format as shown in Table 3.14. The parameters used in the meteorological files are as follows: day number (Day); time of day (Time); wind speed (Wspd in m/s); wind direction (Wdir in °); Anemometer height (AnmHt in m); ambient temperature (AmpT in °C); ambient pressure (AmbP in mBar); cloud cover (Cldc in decas); relative humidity (RelH in %); Wind profile (Wprof, which is a PANEIA switch, where 0 is selected for uniform profile, 1 for power law and 2 for log law); power law exponent if Wprof equals

1, (Pow); vertical temperature profile pattern (Tprof, which is a PANEIA switch, where 0 is selected for uniform profile, 1 for two-step and 2 for log law); temperature lapse rate (LapsRt in °C/min, but only used for two-step profile); inverse temperature lapse rate (Invlaps in °C/min, but only used for two-step profile); mixing height (MixHt in metres, but only used for two-step profile).

Table 3.17 Annotated example of emission data file in PANEIA compliant format

POINT	[point source]
2 1 0 1 224	
240	[number of points]
1	[point number 1]
37	[stack height]
4328 5738	[domain x,y, coordinates (m)]
185.0 378.3 0.00E+00 1.00E+09	[massflow, temp (K), instant of release, duration of release]
3.25E+00 1.83E+01 9.00E+01 0.00E+00	[Stack diameter, exit velocity, angle in x direction, angle in y direction]
0	
63.097 0.000 1.352	[NO_x, PM_{10} and SO_2 concs (μg/g)]
0 0 0 0	[particle parameters – not used]
2	
60	
3361 3771	
135.3 378.3 0.00E+00 1.00E+09	
2.78 18.2 9.00E+01 0.00E+00	
0	
50.470 4.434 0.000	
0 0 0 0	
3	
60	
3292 3771	
135.3 378.3 0.00E+00 1.00E+09	
2.78 18.2 9.00E+01 0.00E+00	
0	
50.470 4.434 0.000	
0 0 0 0	

Table 3.18 Example of part of a PANEIA-compliant meteorological data file

Day Time Wspd Wdir AnmHt AmpT AmbP Cldc RelH Wprof Pow Tprof LapsRt Invlaps MixHt
01022005 0 0.22 125 10 27.1 1013 2 95 2 0 2 0 0 0
01022005 100 0.23 142 10 26.8 1012 2 96 2 0 2 0 0 0
01022005 200 0.41 142 10 26.9 1011 2 96 2 0 2 0 0 0
01022005 300 0.68 140 10 26.9 1010 2 96 2 0 2 0 0 0
01022005 400 0.85 136 10 26.8 1010 2 95 2 0 2 0 0 0
01022005 500 0.99 144 10 27.1 1012 2 95 2 0 2 0 0 0
01022005 600 1.45 146 10 27.7 1013 4 90 2 0 2 0 0 0
01022005 700 1.43 143 10 28 1015 4 88 2 0 2 0 0 0
01022005 800 2.19 147 10 28.9 1018 4 83 2 0 2 0 0 0
01022005 900 2.65 144 10 29.8 1020 3 75 2 0 2 0 0 0
01022005 1000 2.79 144 10 30.5 1021 3 71 2 0 2 0 0 0
01022005 1100 3.28 144 10 30.9 1019 3 68 2 0 2 0 0 0

01022005 1200 3.44 141 10 31.2 1017 2 67 2 0 2 0 0 0
01022005 1300 3.09 145 10 31.3 1015 2 65 2 0 2 0 0 0
01022005 1400 2.72 155 10 31.1 1013 2 65 2 0 2 0 0 0
01022005 1500 2.59 155 10 30.5 1012 2 68 2 0 2 0 0 0
01022005 1600 2.47 150 10 29.9 1011 2 73 2 0 2 0 0 0
01022005 1700 2.09 154 10 29.3 1012 2 77 2 0 2 0 0 0
01022005 1800 2.15 156 10 28.8 1012 1 81 2 0 2 0 0 0
01022005 1900 2.05 150 10 28.7 1013 1 82 2 0 2 0 0 0
01022005 2000 1.58 148 10 28.5 1014 1 85 2 0 2 0 0 0
01022005 2100 1.97 150 10 28.5 1015 4 85 2 0 2 0 0 0
01022005 2200 1.77 149 10 28.3 1015 4 86 2 0 2 0 0 0
01022005 2300 1.96 144 10 28.4 1015 4 87 2 0 2 0 0 0

The modelling studies were carried out only for the year 2005, and used the parameters listed in Table 3.15. A flat terrain was assumed (as for the ADMS studies), and no chemistry model was used, though this is available as part of the model if required. Point sources were used in preference to stack sources because in the latter case PANEIA would attempt to model wind flows around the stacks, greatly increasing modelling time; point sources are suitable for large domain sizes such as for the present model, where wind effects around the stack structures are insignificant compared to other dispersion processes. The wind speed profile was assumed to show a logarithmic increase with height; this profile is corrected for atmospheric stability based on Monin-Obukhov and is commonly used in such studies. Of the alternatives, uniform profile should be used when the vertical extent of the domain is very small, e.g. a few10's of metres, and where the wind has little or no bearing on dispersion, whereas the power-law, with appropriate exponent, is selected when the condition of atmospheric stability is known or when the dispersion has to be studied under specific stability conditions. For the vertical temperature profile the program authors recommend that the log-law profile be used in cases such as this where there is insufficient data on the actual profile; PANEIA automatically computes the profile based on the solar radiation and surface data. For these studies, solar radiation was calculated from the cloud cover, though net solar radiation can be input directly.

There are several turbulence models that can be used in PANEIA. For this study, k-diff was used; the model captures the turbulence in the atmospheric boundary layer and not the mechanical turbulence generated by buildings/obstacles and hilly terrain. The model works well for near-flat terrain without obstacles.

The buoyancy model selected was full gravity, in which gravitational force is computed absolutely without any approximations. This model is the most accurate of the options available, though more computational time is required.

The model layout, including water bodies (blue), urban areas (red), fields (green), emission sources and monitoring points is shown in Figure 3.6. Monitoring points serve the same purpose as in ADMS and ISC, i.e. it allows a time series of predicted concentrations at a specific point to be compiled. The model monitoring points were located at the ambient air quality monitoring stations.

Table 3.19 Parameters used in PANEIA modelling studies

Year	2005
Terrain data	
Longitude (°)	101.2
Latitude (°)	12.35
Roughness length (m)	1.5
Computational domain (m x m x m)	8400(x) x 11100(y) x 1000(z)
Monitor point height (m)	2
Source data	
Type of source	Point sources (entered as emission data file)
Weather data	
Meteorological files	Data emission file
Anemometer height (m)	10
Wind profile	Log
Wind Exponent	Log
Temperature lapse rate	n/a
Temperature mixing height (m)	n/a
Temperature inverse lapse rate	n/a
Simulation options	
Mesh size	18(x) x 18(y) x 16(z)
Grid fineness parameter	10
Fluid type	incompressible
Temperature model	Solve
Wind model	Solve
Buoyancy model	Full gravity
Wall type	No slip
Turbulence model	k-diff

3.10 Summary

This methods section has described the data sources and methodology used in the meteorological and air pollution characterisation studies, health impact analysis, multiple linear regression correlation analysis and atmospheric dispersion modelling analysis. The rationale for the selected methodology has been outlined and the data validity for each of the data sources reviewed.

Chapter 4 outlines the results of the characterisation of meteorology and air quality in the Maptaphut area, and of the statistical analysis of the correlation between pollution and respiratory illness.

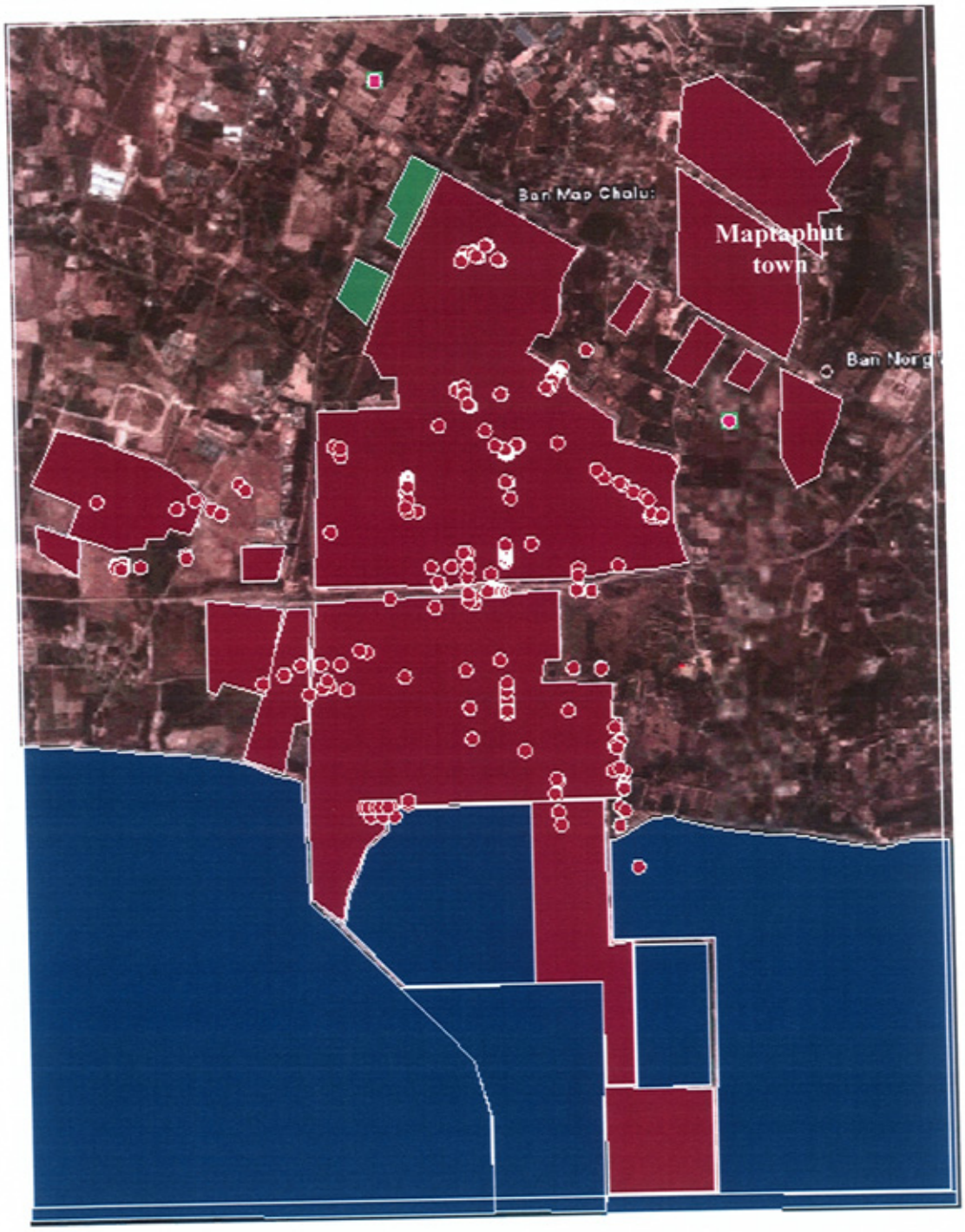


Figure 3.6 PANEIA model showing sources (red circles), monitoring points (pink circles on green squares) and main features.

Chapter 4

Results and Discussion: Characterisation of meteorological, air pollution and health effect data

4.1 Meteorological Data Analysis

Atmospheric conditions can exert a powerful influence on the distribution of pollutant concentrations in space and time. Low wind speeds lead to the build-up of local pollutant concentrations. Strong ventilation with high wind speeds prevents the local build-up near the sources, but contributes to long-range transport and regional pollutants particularly during directional persistent wind conditions.

4.1.1 Monsoon Regime

On the large scale, the climate in Thailand varies seasonally according to the typical monsoon regime. There are traditionally three seasons a year: summer, rainy season, and winter. The wind has a persistent direction in the two seasons: rainy season and winter. In summer, no definite direction of wind exists so the direction can vary. Normally, February to May is the summer; June to September is the rainy season and October to January is the winter. However, the winter may extend into February and the rainy season may start in May.

The SW (South-West) monsoon wind starts in mid May and brings a stream of warm, moist air from the Indian Ocean towards Thailand, causing abundant rain over the country. The NE (North-East) monsoon (winter wind) prevails from October to January with the presence of a high pressure area over China, which extends a ridge to Southeast Asia.

Technically, two major seasons are identified in the country, which are the dry and the wet season. The dry season lasts from mid-October to mid-May. It can be subsequently classified into two periods. The first period, from mid-October to mid-February, is characterized by mild weather of the winter monsoon, which is referred to as winter. The second period, from 16 February to 15 May, is known as local summer in Thailand and is extremely hot (Ostro et al., 1999). The rainy or wet season starts from mid-May and ends

in mid-October. High precipitation and humidity are common throughout the country in this season.

4.1.2 Seasonal Effects on Pollutants

It worth mentioning that the NE wind during the dry season is associated with the extension of a high pressure ridge from southern China to South-East Asia, which is expected to reduce pollutant dispersion. This continental air mass also brings long range transport air pollution to the study area.

During the wet season, air is normally unstable due to the association with the intertropical convergence zone (ITCZ). Wind in the wet season is mostly from Westerly to Southerly direction, bringing relatively clean ocean air masses from the Gulf of Thailand to the area. Wet removal of pollutants further reduces atmospheric pollution, which together results in relatively clean air in the rainy season.

4.1.3 Monthly Wind-Rose

The seasonal change of wind is clearly seen in the monthly wind-roses prepared based on five-year hourly wind data (1998-2002) at 10, 50, and 100m heights of the 100-metre meteorological mast that is located six kilometres away from the coast.

During the dry season, October-December, the NW and NE surface winds predominate (Fig. 4.1). Northerly winds are also observed in January, but in other months of the year the southerly wind component dominates. Wind speeds at ten metres are generally low and calm conditions are observed at high frequency, ranging from 12% in March to 42% in October. For the period from July to December, low winds, less than 1-3 knots (around 0.5-1.5 m/s), are mostly observed in the area. Southerly wind components from January to June have higher speeds, up to 7-10 knots and in some cases up to 17-21 knots are observed, which may be associated with storm conditions. Wind direction at 50 and 100 metres height shows the same pattern, but with higher wind speeds (Fig. A.1 and A.2, Appendix A).

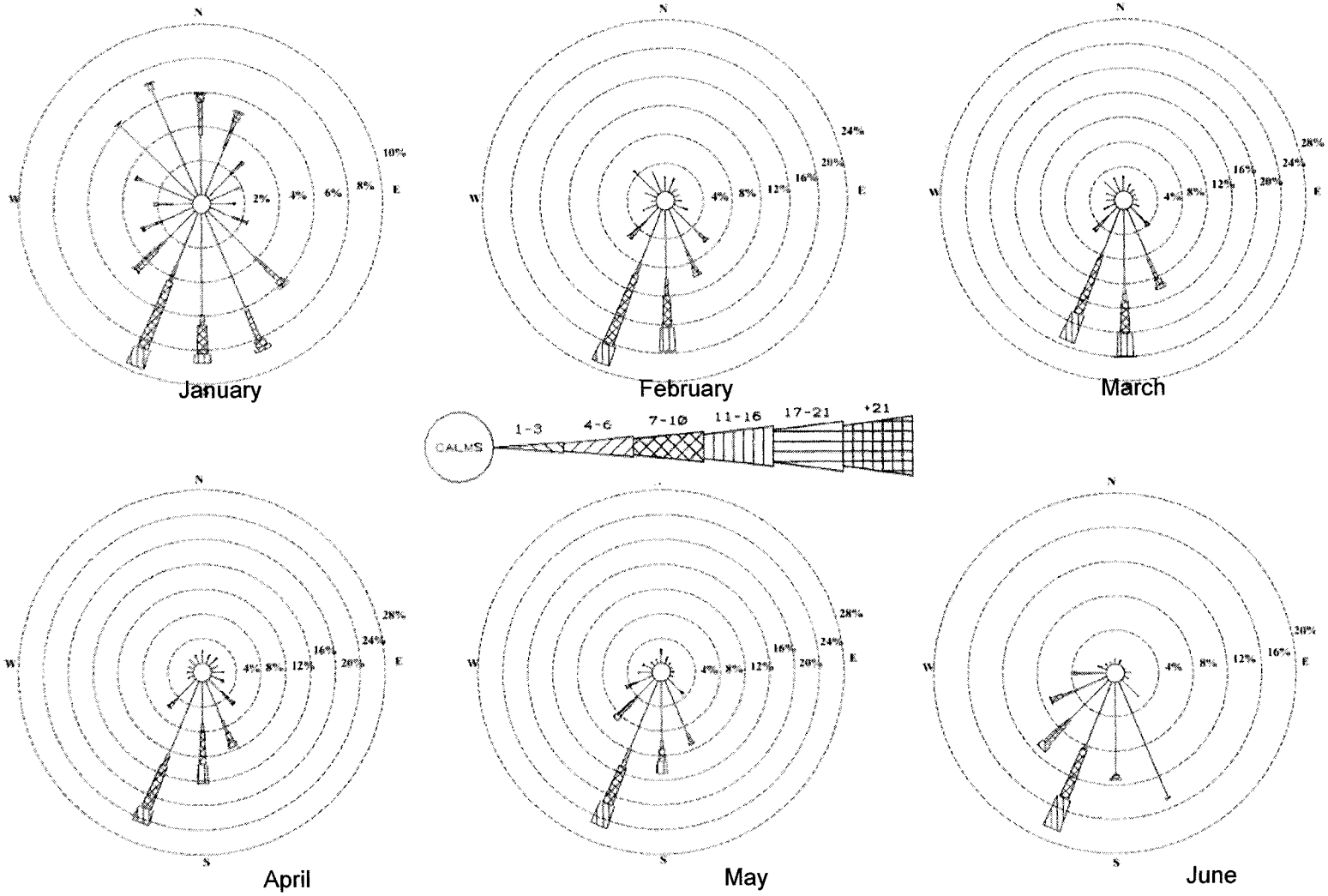


Figure 4.1 Surface wind roses based on 1998-2002 data at 100-metre tall meteorological mast.

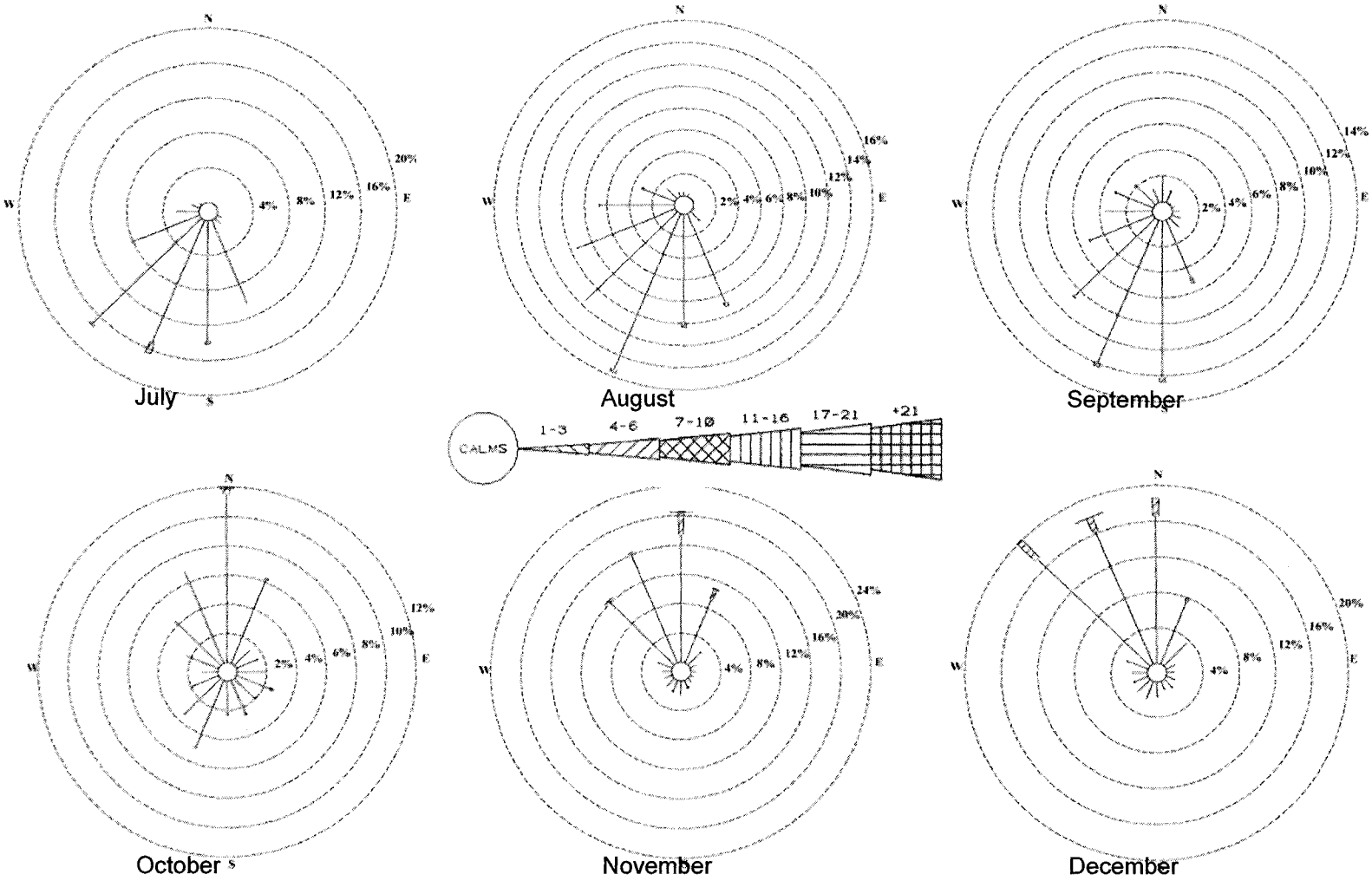


Figure 4.1 Surface wind roses based on 1998-2002 data at 100-metre tall meteorological mast (continued).

4.1.4 Role of Sea-Land Breeze

The coastal line of the Maptaphut area is in the East-West direction. Therefore, the mesoscale sea-land breeze has a South-North orientation. During the daytime, it is from South to North, and during the night-time, the direction is reversed.

In tropical areas, sea-land breeze circulation can extend 50-150 kilometres into the inland and can reach a height of two to three kilometres (Li *et al.*, 1985). Rayong/Maptaphut is a coastal area that experiences more sea-land breeze than Bangkok, which is 40 kilometres away from the Gulf of Thailand. The diurnal change of surface wind speed in Rayong/Maptaphut (Figure 4.2) shows a minimum wind early in the morning (around 7:00), but this minimum is about two times higher than the morning minimum wind in Bangkok (Figure A.4, appendix A). On the other hand, wind in Bangkok (Figure A.3, Appendix A) is stronger especially for the Easterly and Westerly component, as compared to Rayong/Maptaphut, for the period from July to December.

The strong sea breeze effect may be the cause of the low wind in the second half of the year in Rayong/Maptaphut. In the dry season, e.g. December, the regional wind is NE and competes with the Southerly sea breeze and weakens the Northerly wind during daytime. At night, the land breeze (Northerly) combines with the regional NE wind and strengthens the Northerly wind. More pronounced effect of sea-land breeze in Rayong/Maptaphut is perhaps the reason for less fluctuating diurnal wind speed in November and December in Rayong (Figure 4.2) as compared to Bangkok (Figure A.4, appendix A).

Rayong is hot with small temperature fluctuations all year round. Average diurnal maximum temperature is observed at around 13:00 pm (Figure 4.3), which is the highest in March-April (32°C) and the lowest in mid rainy season (29°C). Average diurnal minimum is observed at 7:00 am and the lowest in December-January (23°C).

Dispersion and build-up of pollutants in the study area should be strongly affected by the sea-land breeze circulation. The low Northerly land breeze during night time may not transport the air pollution from inland far enough into the sea. During the daytime, the sea breeze will circulate pollution back to the coastal area. Continuous coastal fumigation may occur and result in a high concentration of pollutants in the inland area. This special

feature of the meteorology in the study area should be taken into account when developing air quality management, especially when applying dispersion modelling, for the area.

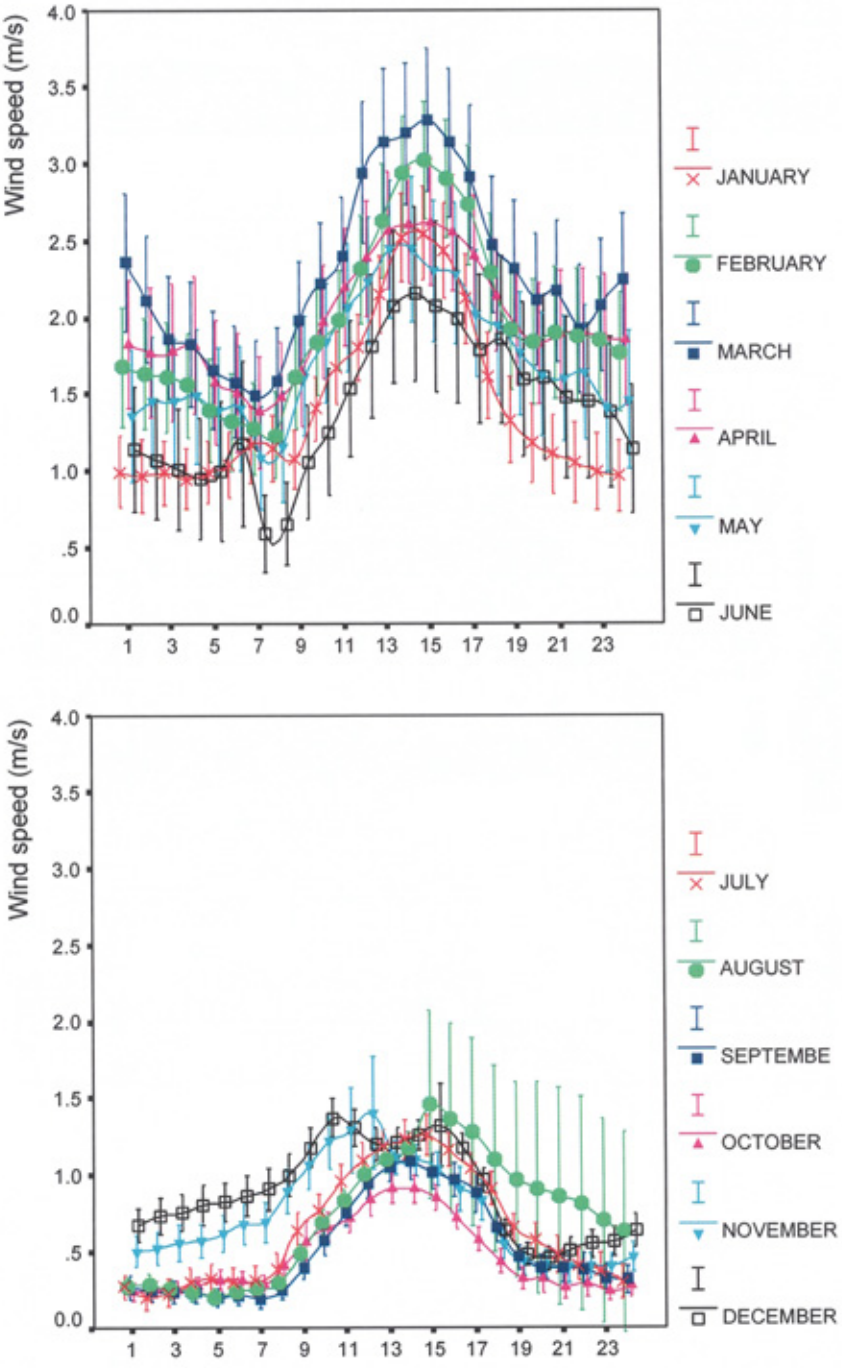


Figure 4.2 Averaged hourly wind speeds observed at 10 m height at 100-metre tall meteorological mast. Vertical bars represent two standard errors of mean values.

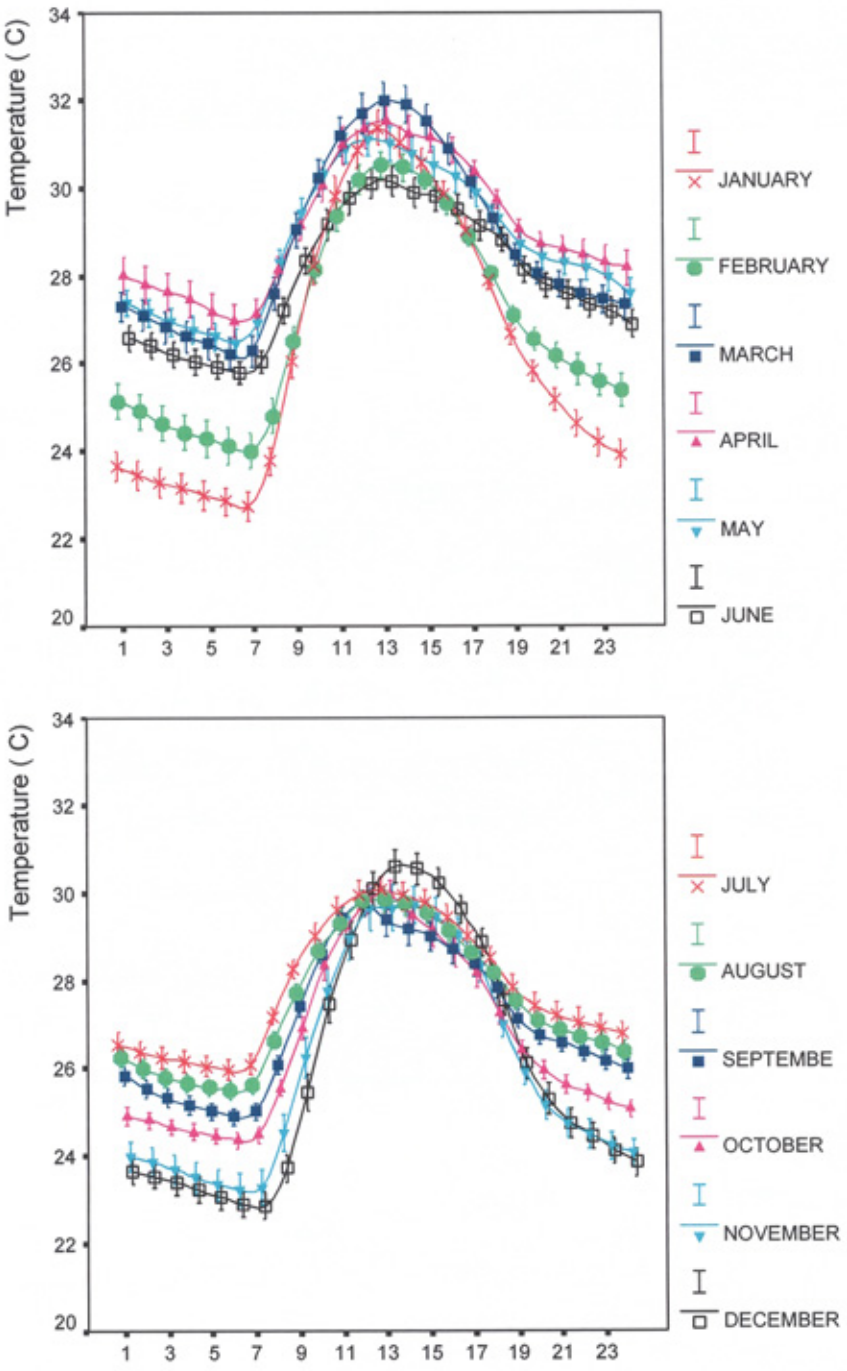


Figure 4.3 Averaged hourly observed surface temperatures at 100-metre tall meteorological mast. Vertical bars represent two standard errors of mean values.

4.2 Status and Trend of Air Pollution in Maptaphut Area

Ambient air quality status and trends have been analysed for a five-year period to determine the air quality characteristics of the Maptaphut Area. The analysed data is compared with Thailand National Ambient Air Quality Standard (NAAQS).

Air quality status and pollution trend of the Maptaphut/Rayong area has been analysed for a period of ten years from 1998 to 2007. These status and trends show the annual characteristics of air pollution in the Maptaphut/Rayong area.

A summary of ambient air quality data during the period of 1998-2007 is shown in Table 4.1. Hourly CO, SO_2, and NO_2 concentrations measured at all three stations during the period never exceeded the Thailand National Ambient Air Quality Standard (NAAQS).

The measured ambient air quality data was compared with national ambient air quality standards by comparing maximum and minimum hourly data with 1-h standard; and comparing average yearly data, based on hourly data, with 1-h, 8-h, or 1-year standard as appropriate. CO concentrations were under the standard at all stations. Some of the maximum O_3 concentrations were over the 1-h standard, but the average concentrations were below the 8-h standard. In 2006, the maximum SO_2 at 29T station exceeded the 1-h standard, though the majority of values were under the standard. The analysis does indicate that PM_{10} seems to be a problem in this area because the average data exceeded the yearly standard in 2000 and 2001. The standard of NO_2 is available only as 1-h standard; this was not exceeded. By way of comparison, the Bangkok air quality data in 2000 gave a 12.8% exceedance frequency for 24-h PM_{10} at curbside stations, and 2.1% at general ambient stations (PCD, 2001). For 1-h O_3 the frequency of exceedances of the NAAQS was 0.3% at ambient stations. None of the Rayong stations would be classified as curbside stations (located within 2-5 m. from main roads), however compared to the ambient stations in Bangkok, air quality in Rayong was more seriously polluted in terms of O_3 and PM_{10}. The 1-h CO NAAQS at curbside stations was exceeded 0.02% of the time in Bangkok, but it was attained for all stations in Rayong.

4.2.1 Trend of Air Pollution in Maptaphut Area

The trend of air pollution including CO, O_3, SO_2, PM_{10}, and NO_2 during the ten year period at stations 29T, 30T, and 31T is shown in Figures 4.4, 4.5 and 4.6, respectively. Figure 4.7

presents the average trend of the three stations. Maximum PM_{10} and O_3 at the stations exceeded the NAAQS. There was not a clear trend of air pollution in the study area during the ten years though there was a slight increase in average PM_{10} and O_3 at stations 29T and 30T from 1999 to 2001.

Maximum PM_{10} ranged from 42 $\mu g\ m^{-3}$ (2005) to 174.8 $\mu g\ m^{-3}$ (2002) at 29T, from 79 $\mu g\ m^{-3}$ (1998) to 195.2 $\mu g\ m^{-3}$ (2001) at 30T, and from 83.6 $\mu g\ m^{-3}$ (1999) to 196.8 $\mu g\ m^{-3}$ (2001) at 31T. Average PM_{10} was in the range of 15.4-59.7 $\mu g\ m^{-3}$ at 29T, 29.5-71.7 $\mu g\ m^{-3}$ at 30T, and 28.4-63.5 $\mu g\ m^{-3}$ at 31T. In terms of average values, station 30T, located in the Rayong town, had the highest PM_{10} pollution followed by station 31 T and the least pollution is station 29T.

Maximum recorded O_3 values were range from 72 ppb (1998) -161 (2004) at 29T, from 88 ppb (2005) to 131 ppb (1999-2000) at 30T, and from 100 ppb (2004) to 176 ppb (2007) at 31T. The average O_3 was in the range of 13-19 ppb at 29T, 14-22 ppb at 30T, and 15-22 ppb at 31T. Both average and maximum values of O_3 at 31T station were the highest among these three stations.

Average O_3 concentration is higher at station 30T and 31T and lower at 29T. From the windroses discussed earlier, the Southerly wind prevails in Rayong; hence, both stations 29T and 31T are located mostly downwind of the industrial estates. However, station 29T is located within two metres from a side road and the titration effects of NO directly emitted from vehicles may result in reduced O_3 at the station. Station 31T is located about 200 metres away from Road 3 and so there may be less NO titration effect and higher O_3 is observed. Station 30T is located inside Rayong town and also is close to a small side road; hence, NO titration effect may reduce O_3 to some extent. To confirm the NO titration effect, daily variation of the ratios of NO to NO_x at the stations (Figure 4.8) was determined. Since only about 5%-10% of emitted NO_x is in the form of NO_2 while the rest is NO, the closer to the emission sources the higher the NO/NO_x ratio. Station 29T has the highest NO/NO_x ratio for most of the time except for the morning rush hour when station 30T has the highest ratio. Station 31T has the lowest NO/NO_x indicating its larger distance from NO_x emission source and explaining higher O_3 observed at the station.

Table 4.1 Ambient air quality in Rayong*

29T Maptaphut	1998		1999		2000		2001		2002	
	Avg	Frequency of exceeding standard (%)	Avg	Frequency of exceeding standard (%)	Avg	Frequency of exceeding standard (%)	Avg	Frequency of exceeding standard (%)	Avg	Frequency of exceeding standard (%)
CO (1-hr) ppm	0.4	0 (0/3053)	0.4	0 (0/5456)	0.5	0 (0/6101)	0.4	0 (0/7362)	0.3	0 (0/8218)
O_3 (1-hr) ppb	13.6	0 (0/4689)	18.8	0 (0/4650)	19.8	0.23 (16/6783)	15.1	0 (0/7588)	14.8	0.01 (1/8180)
SO_2 (1-hr) ppb	2.9	0 (0/4507)	5.4	0 (0/4159)	3.5	0 (0/5486)	4.7	0 (0/6542)	6.1	0 (0/8121)
PM_{10} (24-hr) $\mu g\ m^{-3}$	46.6	3.6 (4/111)	59.7	4.59 (13/283)	54	3.36 (10/297)	48.2	3.43 (9/262)	39.8	2.85 (10/351)
NO_2 (1-hr) ppb	12.7	0 (0/2557)	5.7	0 (0/4550)	13.2	0 (0/7299)	11	0 (0/6721)	10.1	0 (0/8132)
29T Maptaphut	**2003**		**2004**		**2005**		**2006**		**2007 (Jan-Jun)**	
	Avg	Frequency of exceeding standard (%)	Avg	Frequency of exceeding standard (%)	Avg	Frequency of exceeding standard (%)	Avg	Frequency of exceeding standard (%)	Avg	Frequency of exceeding standard (%)
CO (1-hr) ppm	0.4	0 (8061)	0.4	0 (8264)	0.5	0 (8106)	0.5	0 (7926)	0.5	0 (3699)
O_3 (1-hr) ppb	13	0 (0/8038)	13.7	0.05 (4/8097)	19.3	0.04 (3/7791)	16.6	0 (0/7416)	17.4	0.16 (6/3650)
SO_2 (1-hr) ppb	5.6	0 (0/7987)	6.7	0 (0/8096)	5.5	0 (0/7821)	7.5	0.04 (3/7829)	7.9	0 (0/3627)
PM_{10} (24-hr) $\mu g\ m^{-3}$	27.8	0 (0/347)	27.7	0 (0/287)	15.4	0 (0/335)	26.8	0 (0/348)	32.5	0 (0/154)
NO_2 (1-hr) ppb	11.1	0 (8094)	12.3	0 (7970)	9.5	0 (8123)	14.2	0 (7805)	14.6	0 (3692)

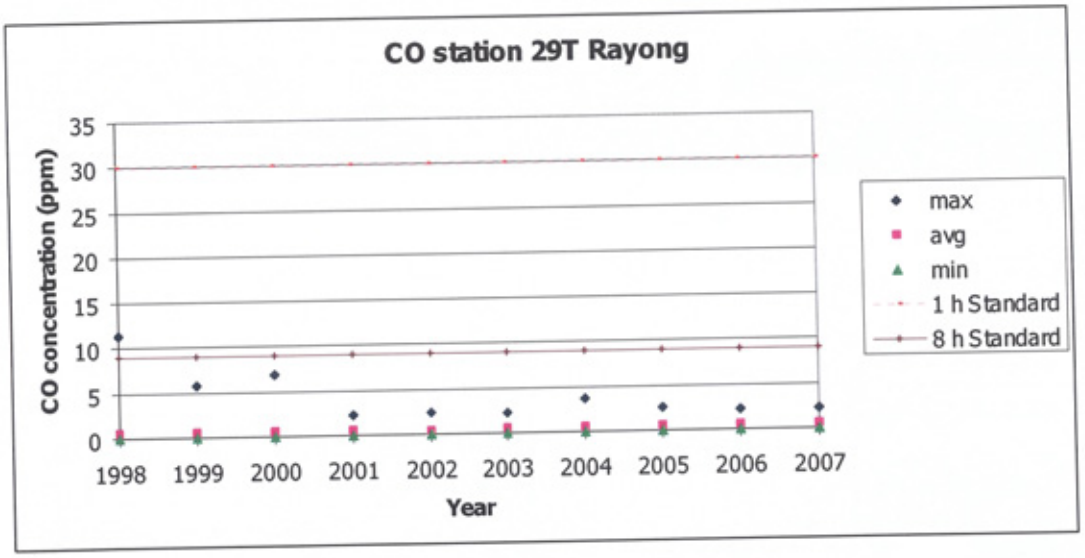
Table 4.1 Ambient air quality in Rayong (continued) *

30T Rayong	1998		1999		2000		2001		2002	
	Avg	Frequency of exceeding standard (%)	Avg	Frequency of exceeding standard (%)	Avg	Frequency of exceeding standard (%)	Avg	Frequency of exceeding standard (%)	Avg	Frequency of exceeding standard (%)
CO (1-hr) ppm	2.2	0.02 (1/5093)	0.5	0 (0/4959)	0.5	0 (0/6219)	0.9	0 (0/6292)	0.6	0 (0/7815)
O_3 (1-hr) ppb	18	0.01 (1/7467)	14.4	0.16 (10/6304)	16.5	0.34 (21/6131)	20.6	0.03 (2/7485)	16	0 (0/7802)
SO_2 (1-hr) ppb	3.4	0 (0/6606)	2.4	0 (0/6127)	4.4	0 (0/5401)	4.2	0 (0/5216)	3.2	0 (0/7733)
PM_{10} (24-hr) $\mu g\ m^{-3}$	29.5	0 (0/160)	38.2	1.36 (2/147)	42.6	2.46 (5/203)	66	10.3 (30/291)	71.7	4.76 (5/105)
NO_2 (1-hr) ppb	10.3	0 (0/5586)	8.9	0 (0/4432)	11.3	0 (0/6489)	10.7	0 (0/7235)	10.2	0 (0/7770)
30T Rayong	**2003**		**2004**		**2005**		**2006**		**2007 (Jan-Jun)**	
	Avg	Frequency of exceeding standard (%)	Avg	Frequency of exceeding standard (%)	Avg	Frequency of exceeding standard (%)	Avg	Frequency of exceeding standard (%)	Avg	Frequency of exceeding standard (%)
CO (1-hr) ppm	0.6	0 (0/8311)	0.6	0 (0/8024)	0.5	0 (0/8123)	0.5	0 (0/8345)	0.5	0 (0/3926)
O_3 (1-hr) ppb	16.9	0.02 (2/8140)	17.1	0 (0/8167)	16.1	0 (0/8335)	17.1	0 (0/8355)	22.6	0.54 (21/3896)
SO_2 (1-hr) ppb	2.6	0 (0/8166)	3.5	0 (0/7721)	3.3	0 (0/8283)	2.9	0 (0/8197)	3.2	0 (0/2991)
PM_{10} (24-hr) $\mu g\ m^{-3}$	43.2	1.08 (3/278)	44.2	3.3 (11/333)	44.3	2.56 (9/352)	38.7	0.85 (3/355)	51.9	6.11 (8/131)
NO_2 (1-hr) ppb	10.3	0 (0/8226)	11.5	0 (0/8137)	10.2	0 (8359)	9.5	0 (8249)	10	0 (3808)

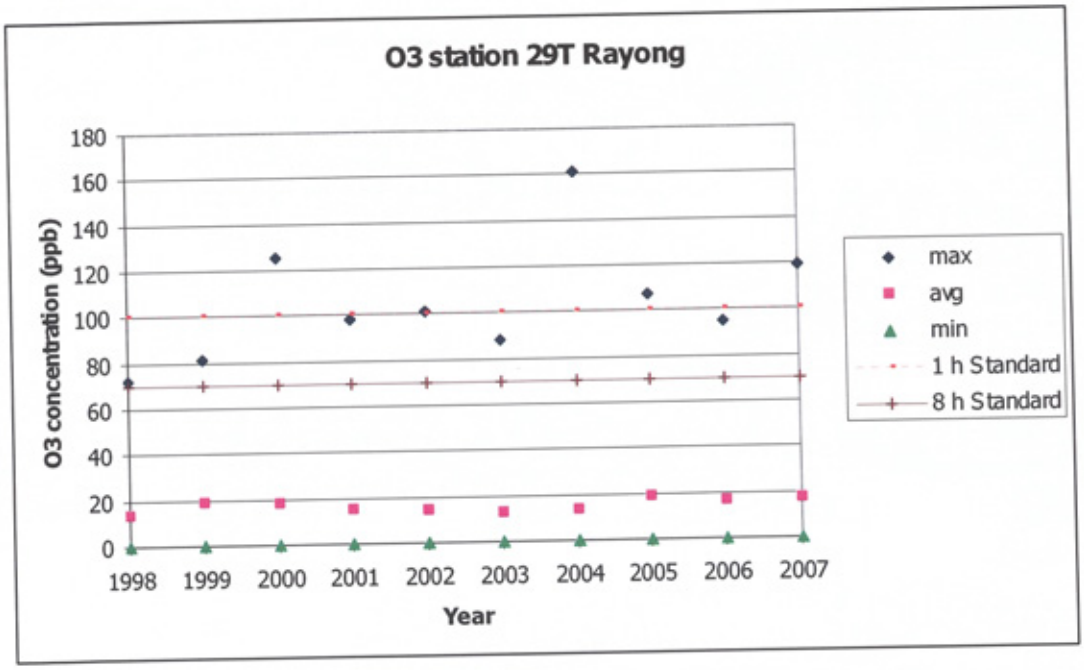
Table 4.1 Ambient air quality in Rayong (continued)*

31T Rayong	1998		1999		2000		2001		2002	
	Avg	Frequency of exceeding standard (%)	Avg	Frequency of exceeding standard (%)	Avg	Frequency of exceeding standard (%)	Avg	Frequency of exceeding standard (%)	Avg	Frequency of exceeding standard (%)
CO (1-hr) ppm	0.5	0 (0/2378)	0.5	0 (0/4319)	0.3	0 (0/6026)	0.4	0 (0/7902)	0.3	0 (0/8221)
O_3 (1-hr) ppb	15.4	0 (0/1660)	22.3	0.27 (4/1474)	18.6	0.16 (11/6779)	16.7	0.05 (4/8071)	16	0.01 (1/8227)
SO_2 (1-hr) ppb	4.8	0 (0/4300)	5.3	0 (0/2938)	5.2	0 (0/6286)	3.4	0 (0/6187)	3.3	0 (0/7622)
PM_{10} (24-hr) $\mu g\ m^{-3}$	29.5	0 (0/71)	33.5	0 (0/46)	63.5	7.14 (11/154)	53.8	3.01 (5/166)	28.4	0 (0/339)
NO_2 (1-hr) ppb	7.4	0 (0/2557)	8.4	0 (0/2377)	7	0 (0/6756)	8.9	0 (0/8091)	8.6	0 (0/8089)
31T Rayong	**2003**		**2004**		**2005**		**2006**		**2007 (Jan-Jun)**	
	Avg	Frequency of exceeding standard (%)	Avg	Frequency of exceeding standard (%)	Avg	Frequency of exceeding standard (%)	Avg	Frequency of exceeding standard (%)	Avg	Frequency of exceeding standard (%)
CO (1-hr) ppm	0.3	0 (0/8101)	0.3	0 (0/8193)	0.3	0 (0/7669)	0.4	0 (0/7972)	0.3	0 (0/3149)
O_3 (1-hr) ppb	18.7	0.06 (5/7724)	19.1	0 (0/7516)	17	0.04 (3/7736)	18.7	0.05 (4/8016)	19.8	0.22 (7/3148)
SO_2 (1-hr) ppb	3.4	0 (0/7895)	2.4	0 (0/8087)	4.1	0 (0/7631)	4	0 (0/7867)	4.4	0 (0/3899)
PM_{10} (24-hr) $\mu g\ m^{-3}$	36.3	1.29 (4/310)	39.2	1.80 (6/334)	30.4	0 (0/232)	50.7	0.93 (3/322)	47.2	0.62 (1/162)
NO_2 (1-hr) ppb	8.9	0 (0/8082)	8.3	0 (0/8173)	7.1	0 (0/7701)	10.8	0 (0/8008)	7.5	0 (0/3779)

* National ambient air quality standards as follows:
CO (1-hr) 30 ppm, CO (8-hr) 9 ppm, O_3 (1-hr) 100 ppb, O3 (8-hr) 70 ppb, SO2 (1-hr) 300 ppb, SO_2 (1-y) 40 ppb, PM_{10} (24-hr) 120 $\mu g\ m^{-3}$, PM_{10} (1-y) 50 $\mu g\ m^{-3}$and NO_2 170 ppb.

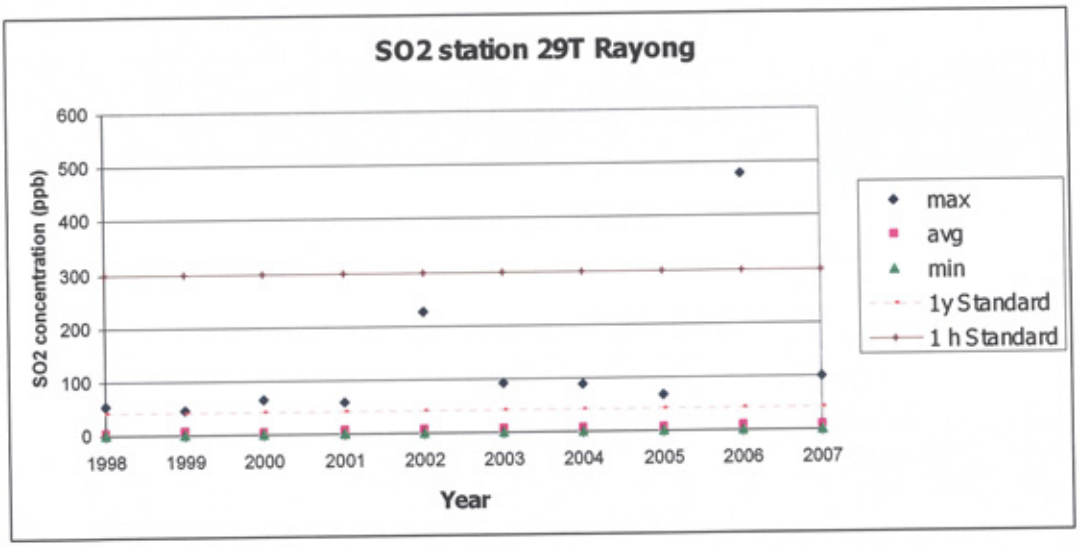


(a)



(b)

Figure 4.4 Pollution trend of (a) CO, (b) O_3, (c) SO_2, (d) PM_{10}, and (e) NO_2 at 29T Maptaphut station.



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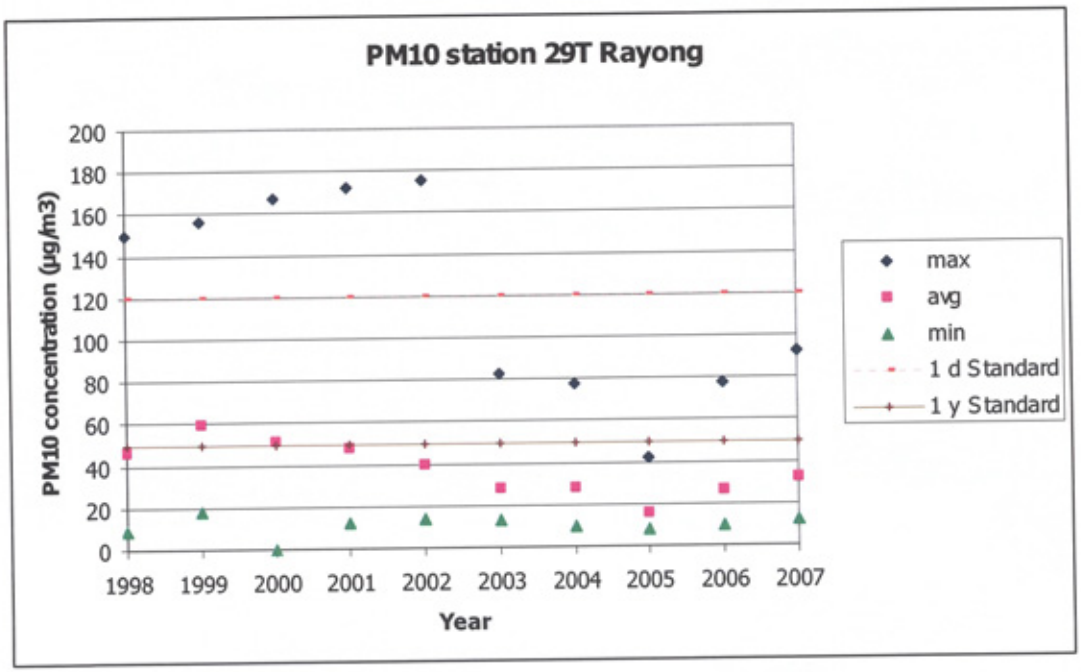
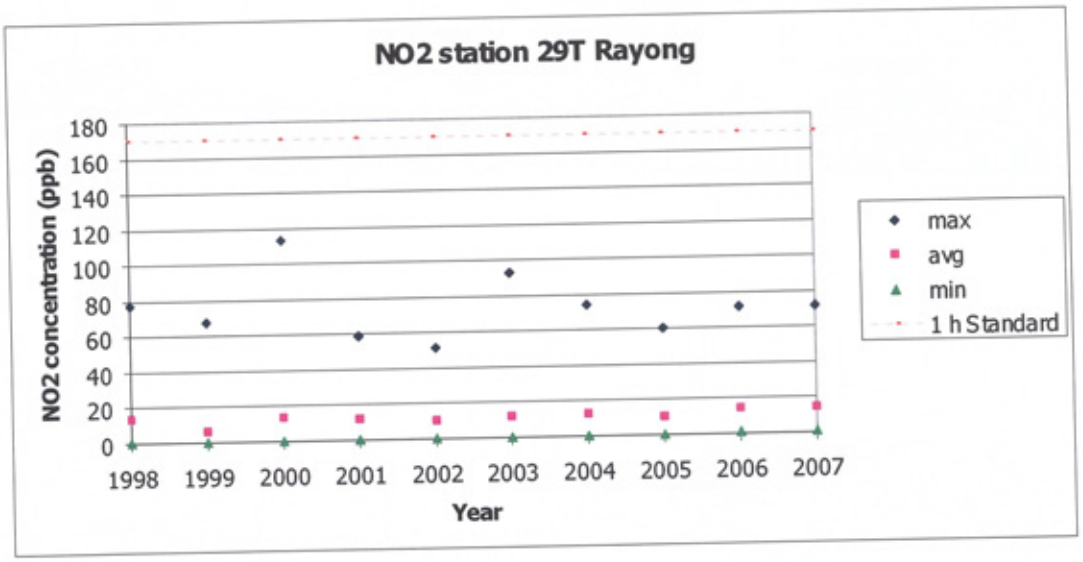


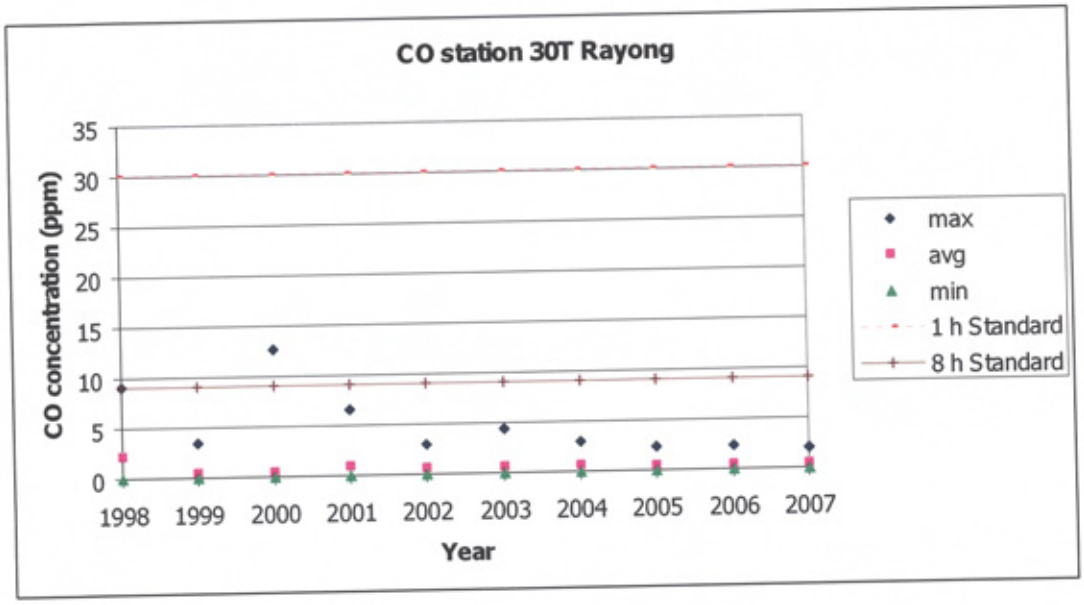
(d)

Figure 4.4 Pollution trend of (a) CO, (b) O_3, (c) SO_2, (d) PM_{10}, and (e) NO_2 at 29T Maptaphut station (continued).



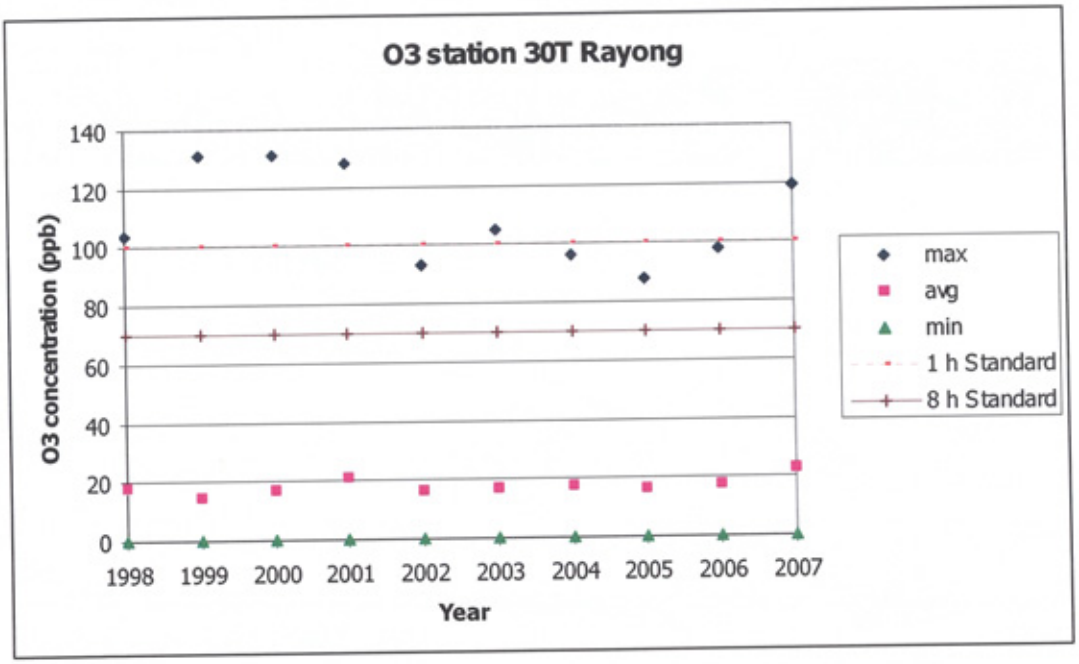
(e)

Figure 4.4 Pollution trend of (a) CO, (b) O_3, (c) SO_2, (d) PM_{10}, and (e) NO_2 at 29T Maptaphut station (continued).



(a)

Figure 4.5 Pollution trend of (a) CO, (b) O_3, (c) SO_2, (d) PM_{10}, and (e) NO_2 at 30T Rayong station.



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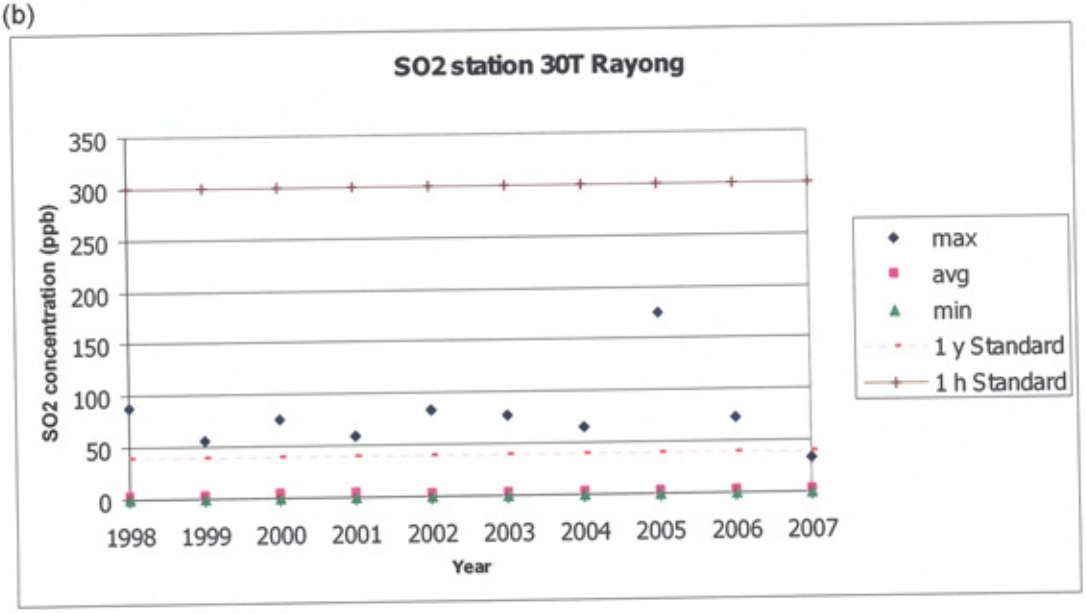
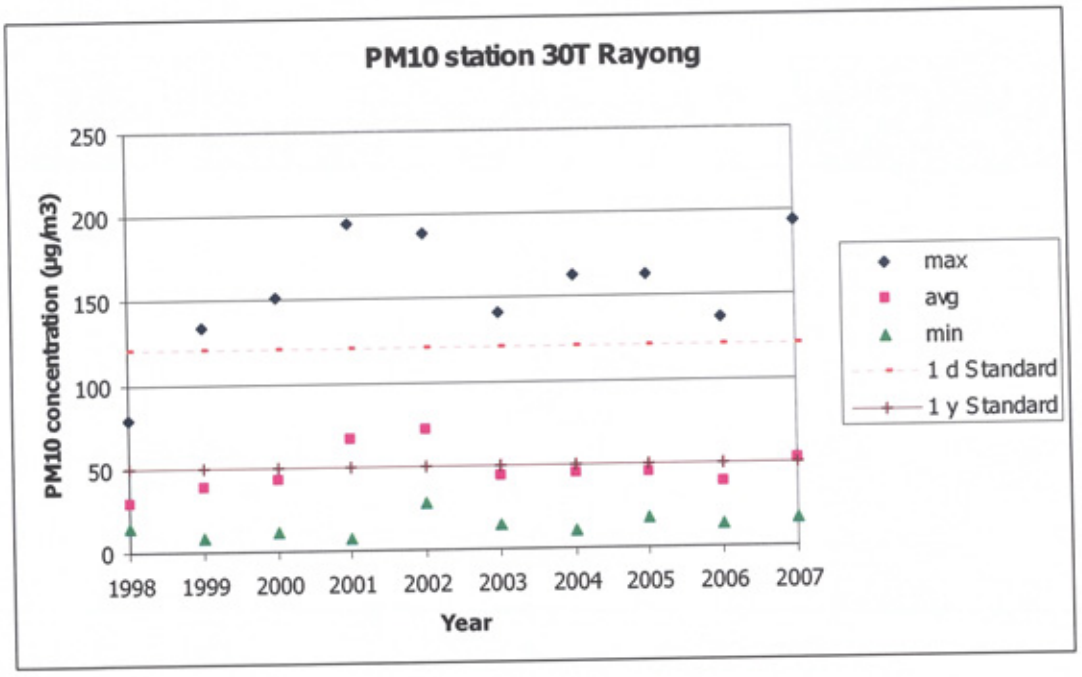
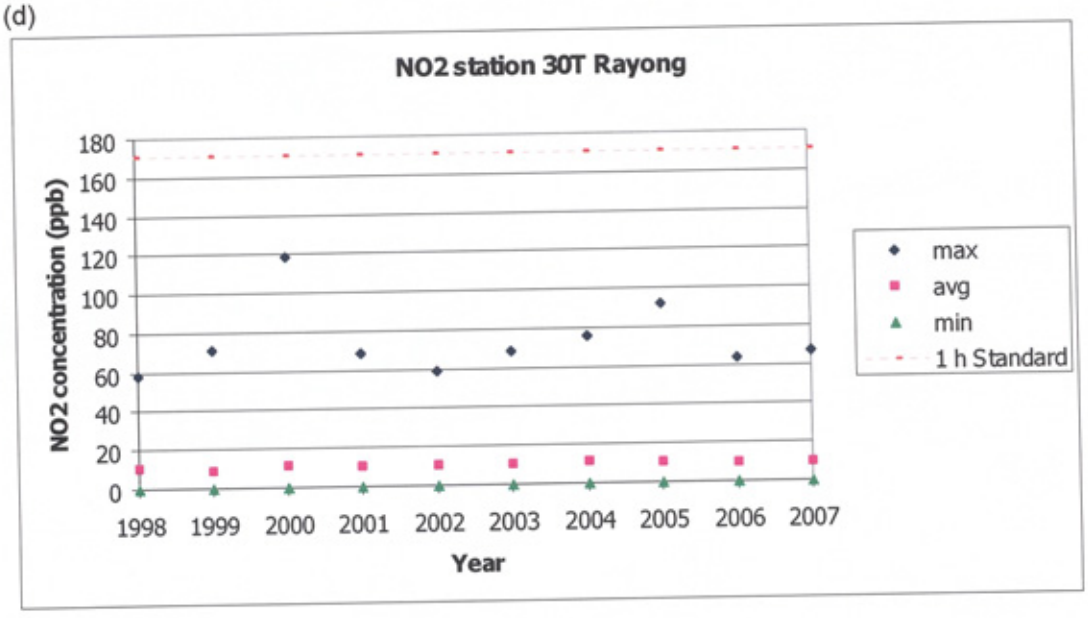


(c)

Figure 4.5 Pollution trend of (a) CO, (b) O_3, (c) SO_2, (d) PM_{10}, and (e) NO_2 at 30T Rayong station (continued).

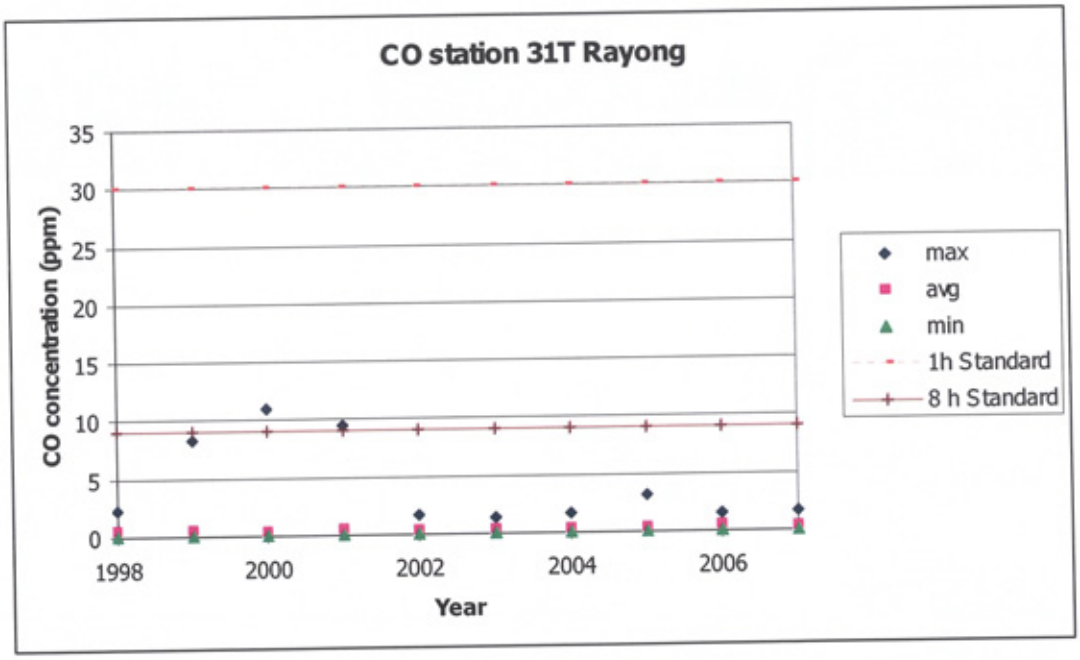


(d)

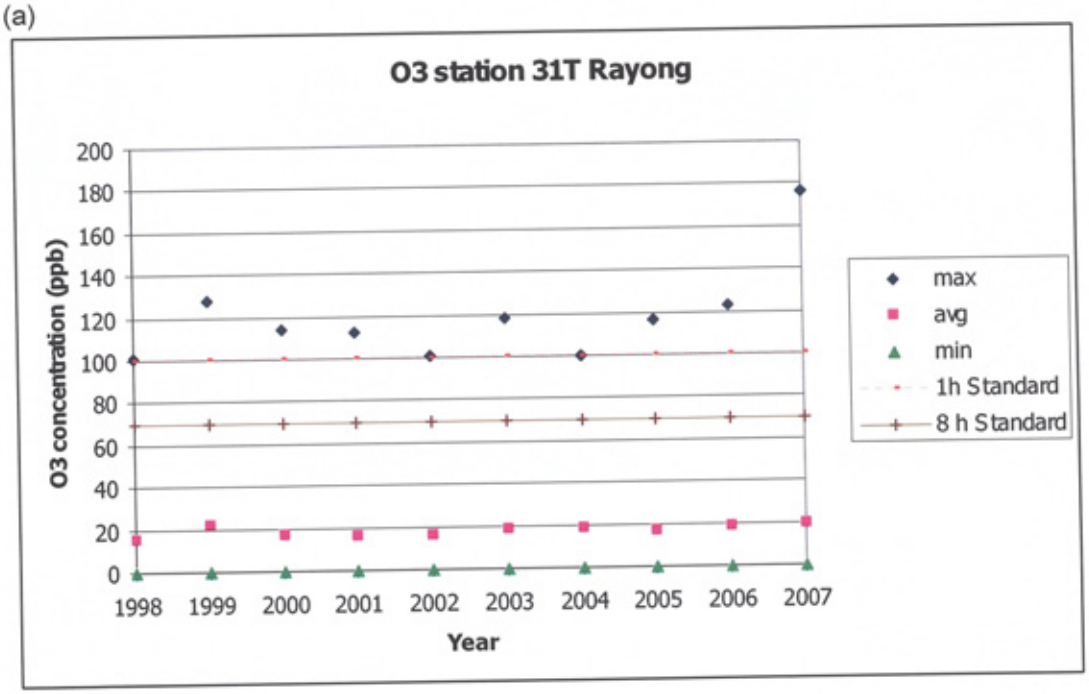


(e)

Figure 4.5 Pollution trend of (a) CO, (b) O_3, (c) SO_2, (d) PM_{10}, and (e) NO_2 at 30T Rayong station (continued).

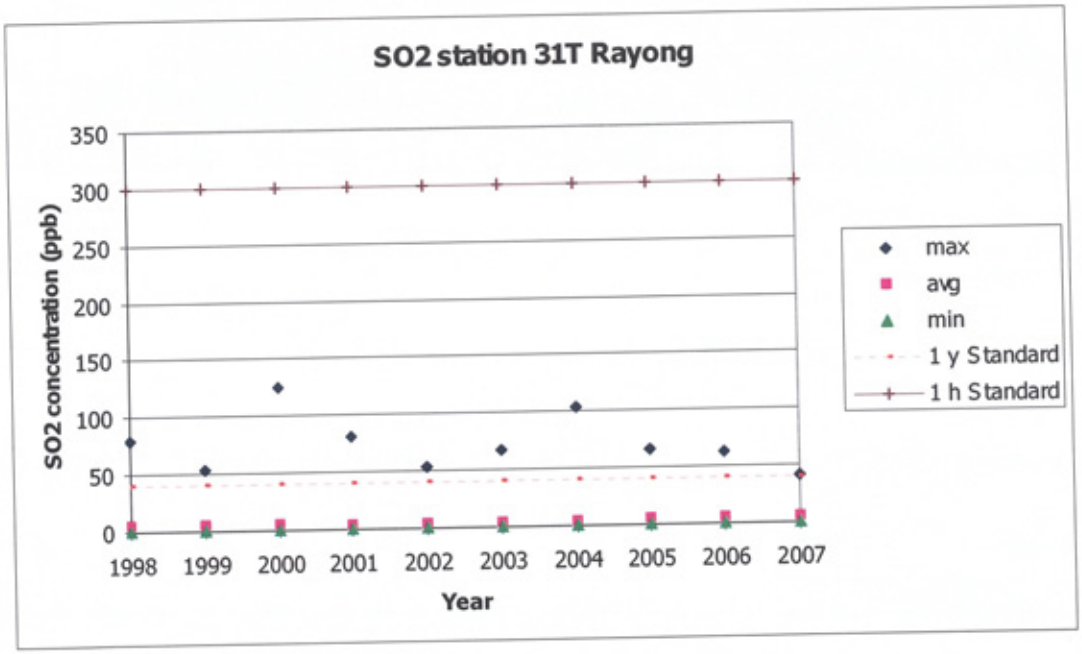


(a)



(b)

Figure 4.6 Pollution trend of (a) CO, (b) O_3, (c) SO_2, (d) PM_{10}, and (e) NO_2 at 31T Rayong station.



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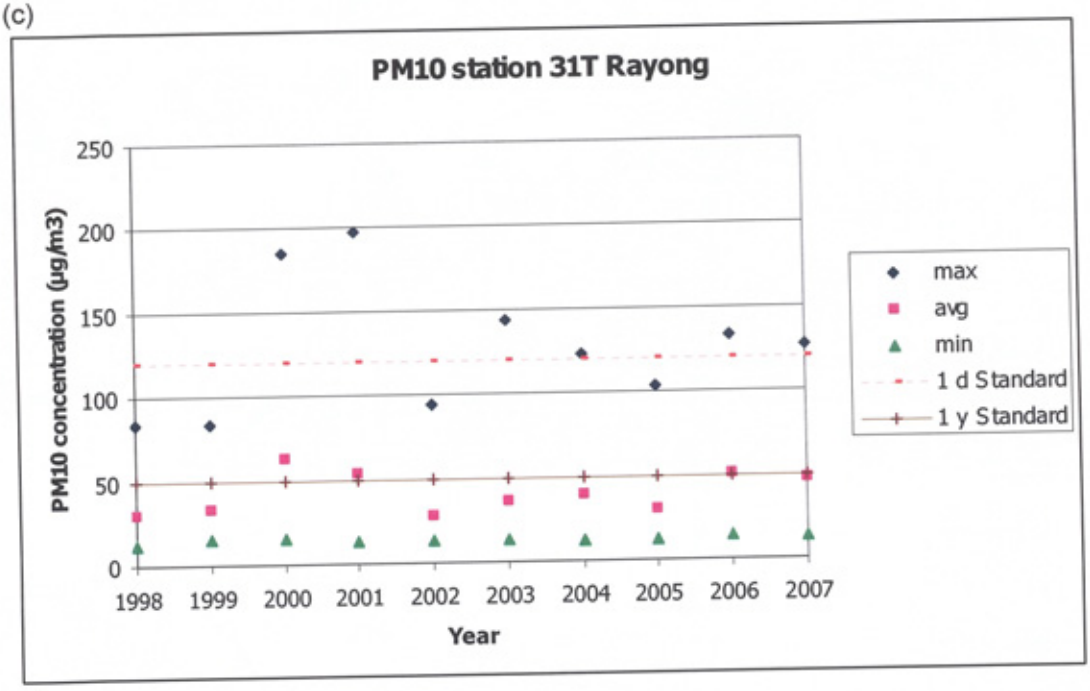
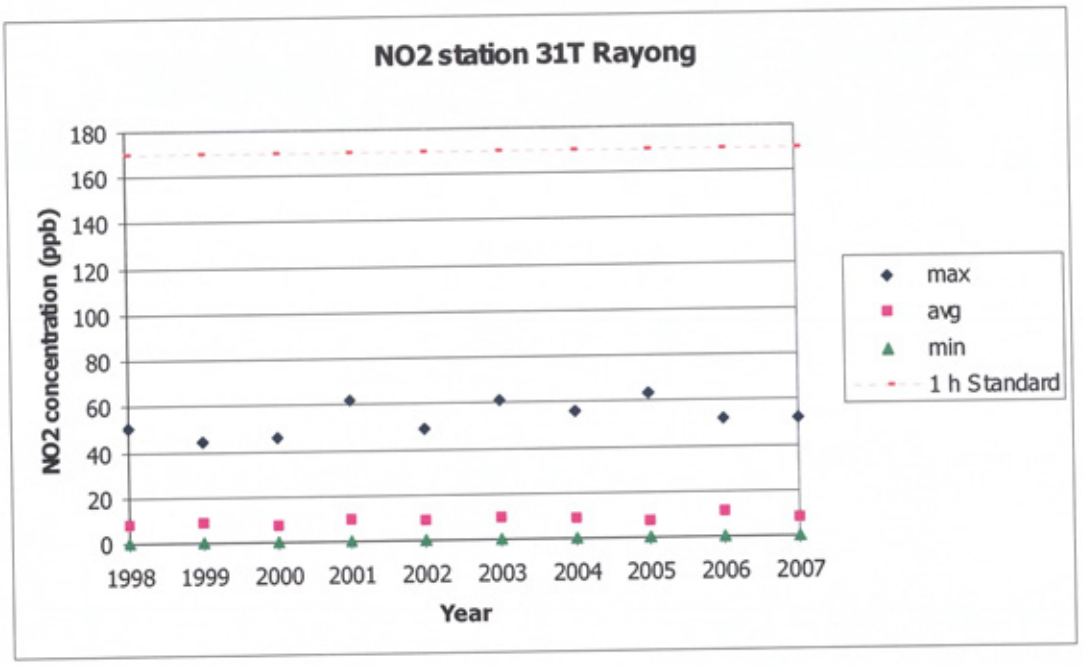


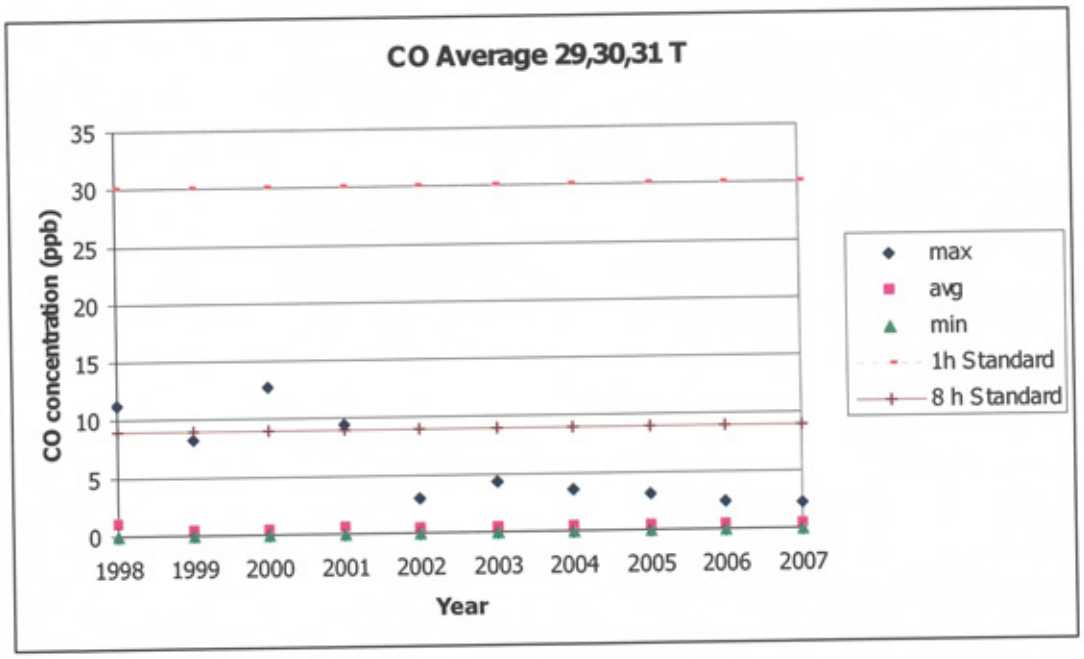
(d)

Figure 4.6 Pollution trend of (a) CO, (b) O_3, (c) SO_2, (d) PM_{10}, and (e) NO_2 at 31T Rayong station (continued).



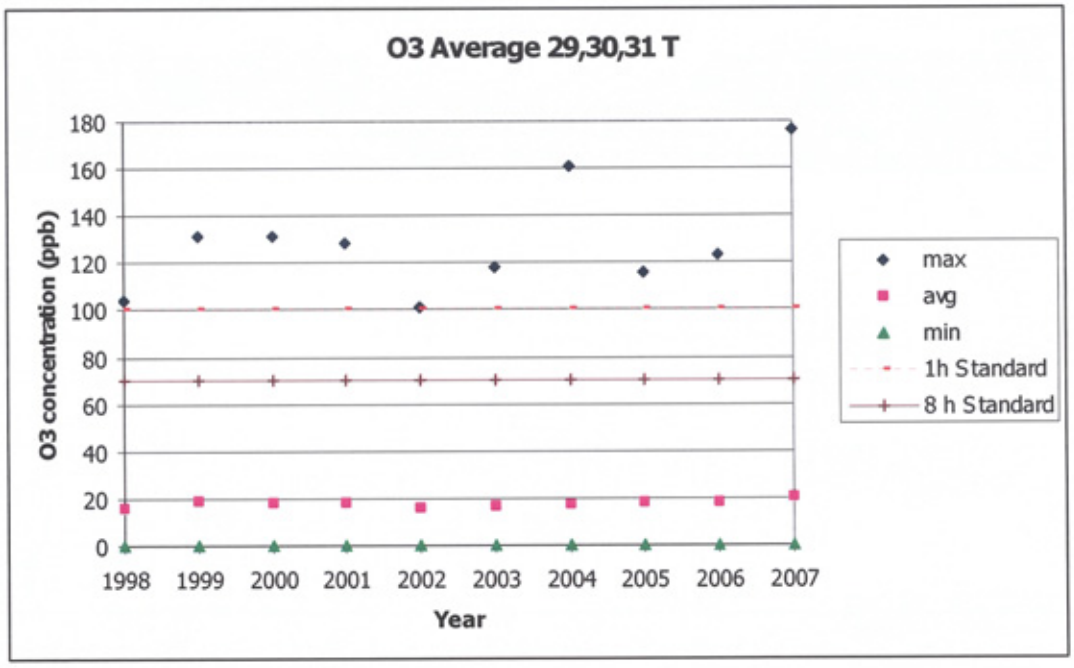
(e)

Figure 4.6 Pollution trend of (a) CO, (b) O_3, (c) SO_2, (d) PM_{10}, and (e) NO_2 at 31T Rayong station (continued).

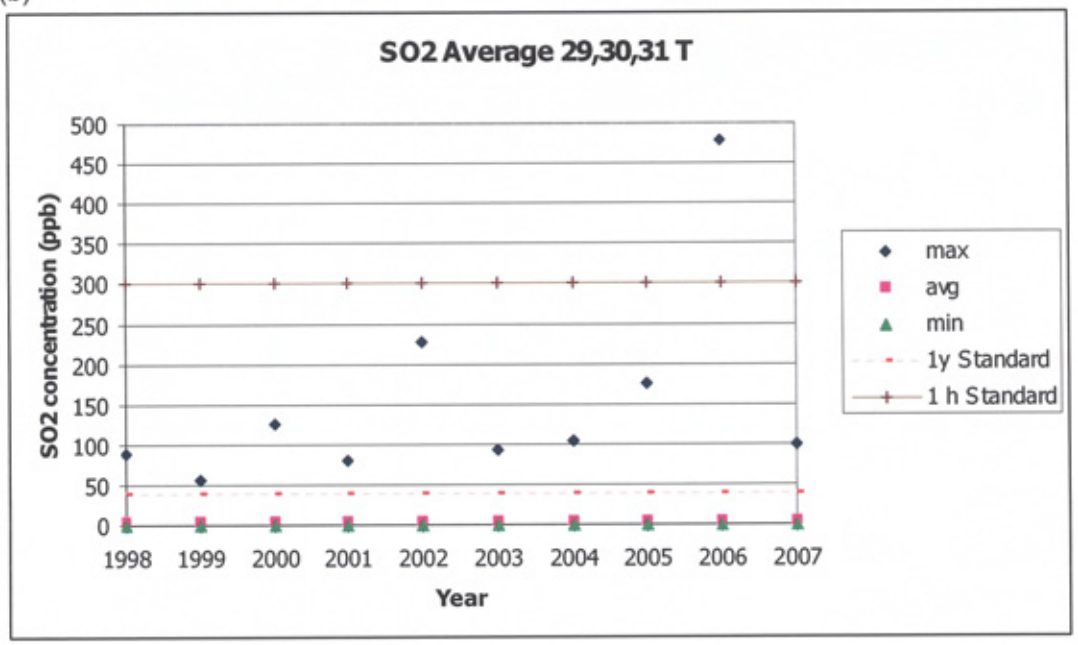


(a)

Figure 4.7 Pollution trend of (a) CO, (b) O_3, (c) SO_2, (d) PM_{10}, and (e) NO_2 in Rayong province (average of observed data at 29T, 30T, and 31T).

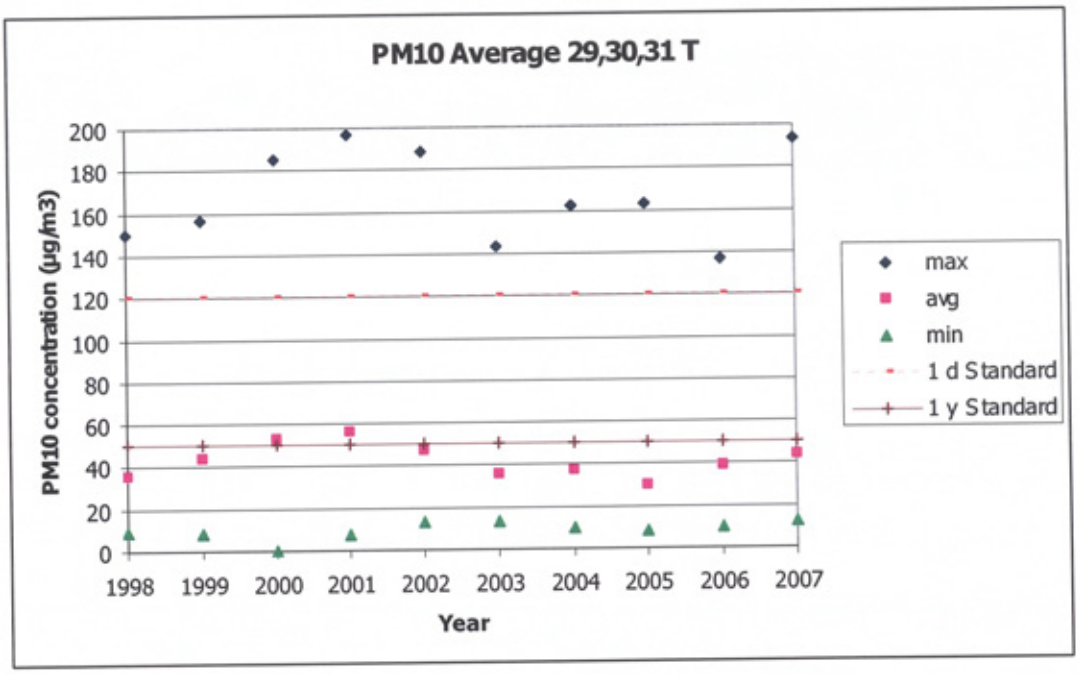


(b)

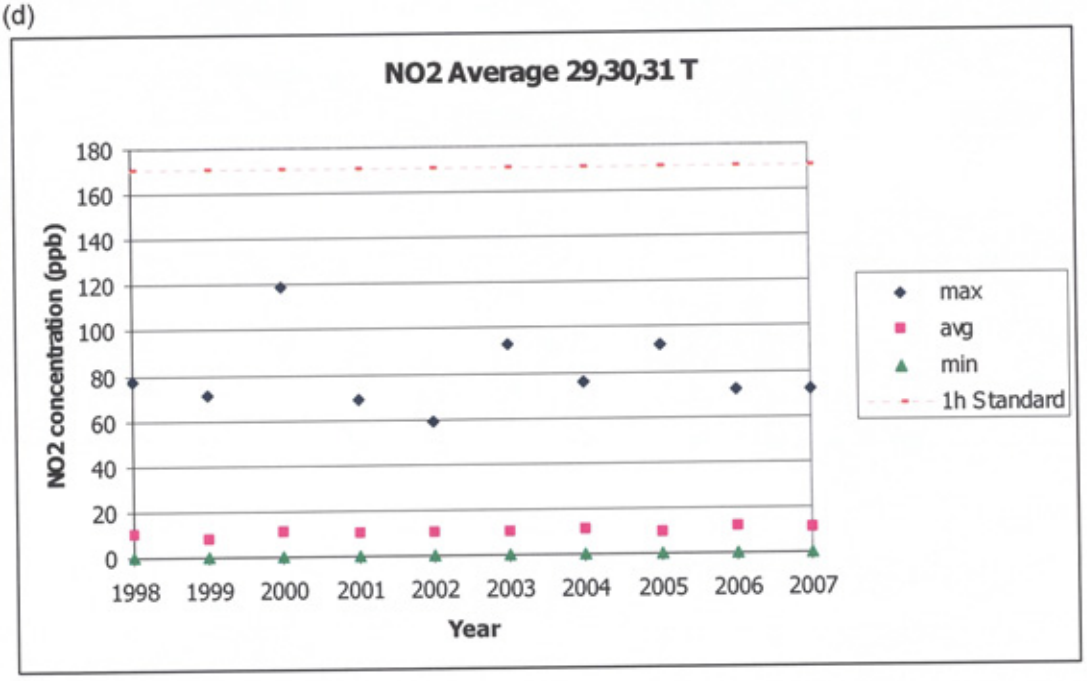


(c)

Figure 4.7 Pollution trend of (a) CO, (b) O_3, (c) SO_2, (d) PM_{10}, and (e) NO_2 in Rayong province (average of observed data at 29T, 30T, and 31T) (continued).



(d)



(e)

Figure 4.7 Pollution trend of (a) CO, (b) O_3, (c) SO_2, (d) PM_{10}, and (e) NO_2 in Rayong province (average of observed data at 29T, 30T, and 31T) (continued).

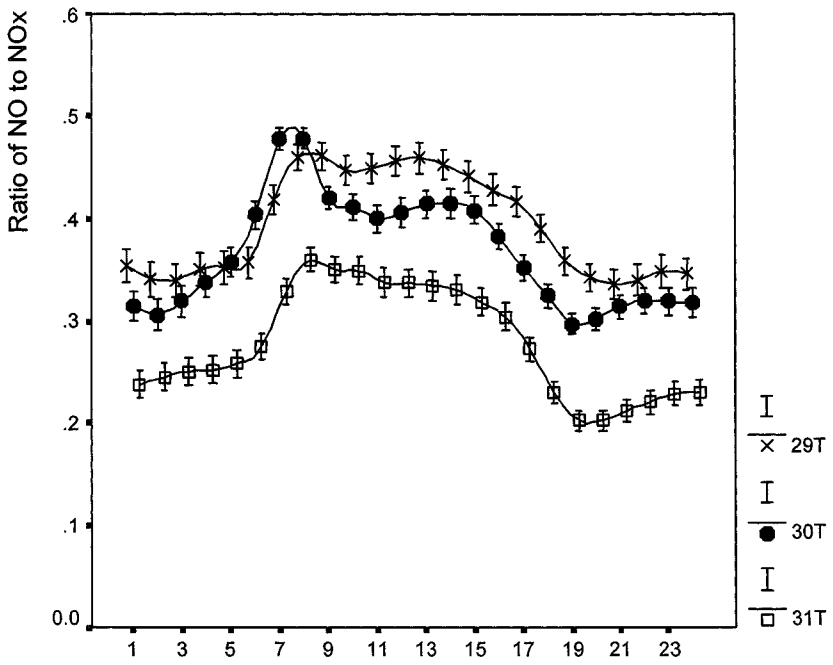


Figure 4.8 Daily average of the ratio of NO to NOx at station 29T, 30T, and 31T based on the data 1998-2002. Vertical bars represent two standard errors of mean values.

4.3 Diurnal Air Quality Status

The diurnal air quality trends for the Maptaphut/Rayong area was analysed for a period of five years from 1998 to 2002. The analysis shows that the diurnal trends are strongly influenced by the meteorological conditions.

Averaged diurnal variations of hourly pollutants' concentrations for each month of the five consecutive years are calculated and then are again averaged to obtain five year averaged monthly diurnal variation.

Averaged diurnal variations of hourly concentration NOx ($NO+NO_2$), O_3, SO_2, PM_{10}, CO, and NMHC (Non-methane hydrocarbons) for each month of the year for three stations are shown in Figure 4.9, Figure 4.10, and Figure 4.11, respectively. Each point in these figures represents the average of up to 155 hourly measurements during the five years.

It is observed that during the wet season and transitional period from April to September inclusively, the study area is not likely to have an air quality problem. The monthly average

diurnal variations of all recorded parameters during the periods are well below the NAAQS.

Diurnal variations of pollutants in the dry season (October to March, inclusively) represent the typical characteristics of polluted areas (Wark et al., 1998) and are quite similar to the variations recorded in Bangkok city (Figure A.5, Appendix A). Morning and evening rush hours (around 8:00 am and 7:00 pm) are characterized with high peaks of NOx, PM_{10}, CO, and NMHC concentrations due to rush hour traffic emission.

Two air pollution peaks corresponding to the morning and evening rush hours can be distinguished while during midday the concentration level drops down significantly. It can be seen from the figures that the decrease in NOx, PM_{10}, CO, and NHMC concentrations at midday is relatively higher compared with the decrease in traffic (Figure 4.12 and 4.13) and consequently the emission intensity (Figure 4.9, 4.10 and 4.11). It can be concluded that a large part of the concentration decreases at midday occurs due to intensive mixing of the atmosphere (Figure 4.9-4.11).

O_3 increases and reaches its maximum at around 13:00. O_3 precursors, i.e. NOx, NMHC, and CO are consumed in photochemical reactions and are minimum when O_3 is maximum. Enhanced ventilation (increased mixing height and wind speed) in the afternoon also contributes to the reduction of the pollutants concentration. The amount of NO_2 in the ambient air depends on its precursors, NO and O_3. Short-time dynamics of NO_2 is a highly nonlinear process restricted by the presence of sufficient amounts of O_3, which vary both annually, and depending on the diurnal solar insolation cycle. The afternoon high level of O_3 occurs partially due to ultraviolet radiation, which reaches the ground level when transport peak ozone decreases due to large amounts of NO.

The SO_2 concentration is low and does not change significantly during the day, probably due to the fact that area and industrial emissions are important sources of SO_2 in Rayong and they fluctuate less than mobile source emissions.

Analysis of diurnal variations of pollution at each station separately shows that the morning peaks, especially PM_{10} and NO_x, are more clearly shown for station 29T and 30T than station 31T. This is explained by the location of stations 29T and 30T, which are closer to traffic sources than 31T as mentioned earlier. The O_3 concentration at station 31T is the highest and the pollution during December, January and February maybe of concern.

At station 29T in December, NO_x pollution may be the main problem although no violation of NAAQS was recorded. The high NO_x value is observed during evening peak hours with an average around 80 ppb and a trend (Figure 4.9) showing NO_2 approaching NAAQS. At 30T, PM_{10} is more serious with an average morning peak at 130 $\mu g\ m^{-3}$, which is most probably from the heavy traffic in town.

The pollutant concentrations in Rayong province are lower than in Bangkok, which is probably due to large traffic sources contributing to the latter. For example, NO_x, PM_{10}, and CO at Singha (downwind of the Bangkok centre during NE monsoon) are 3, 2.0 and 3.6 times higher than at station 29T in January, and are 1.8, 2.5 and 3.5 times higher in August. Peak O_3 concentrations in Bangkok are also higher than in Rayong province, i.e. 40-50 ppb in Bangkok vs. 30-45ppb in Rayong province in January, and 20-25 ppb in Bangkok vs. 10-20 ppb in Rayong in August.

There are a number of chemical factories in Rayong province, especially in the Maptaphut Industrial Estate; hence, NMHC pollution is expected. However, the NMHC concentration recorded is still lower than Bangkok (Figure A.9, Appendix A). In January, the average morning peak of NMHC was 0.8 ppm at 29T and 30T and 0.5 ppm at 31T while it was 1.5 ppm at Huaikhwang (Bangkok centre) and 2.5 ppm at Singha (mentioned above). In August, the morning peak NMHC was 0.3-0.8 ppm in Rayong as compared to 1.8 ppm at Huaikhwang and 1.5 ppm at Singha of Bangkok.

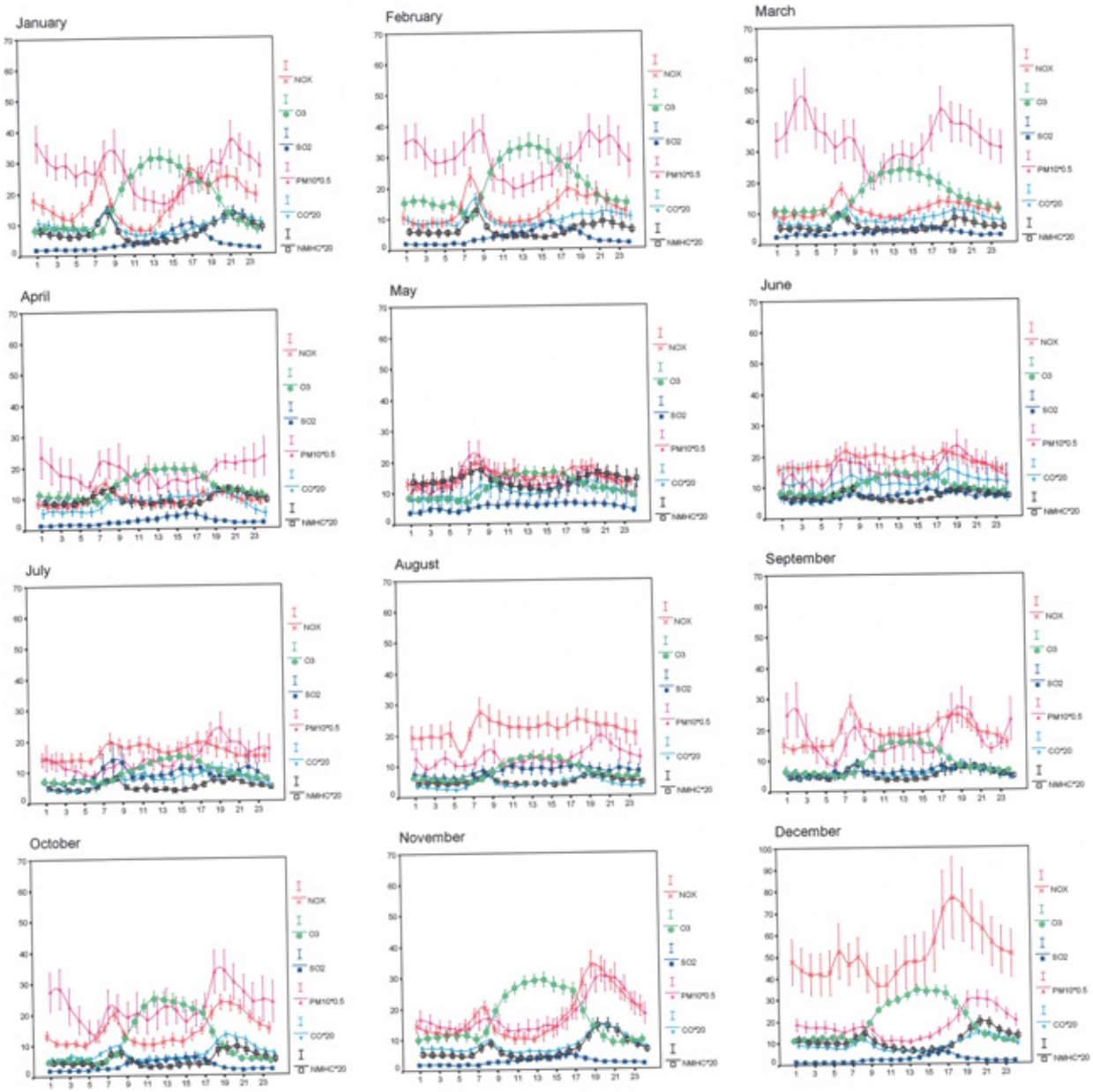


Figure 4.9 Daily variations of pollutant concentrations in each month at 29T station. Note: Vertical bars represent two standard errors of mean values. The values for NO_x, O_3, and SO_2 are in ppb. To obtain PM_{10} values ($\mu g\ m^{-3}$), the readings from graph should be multiplied by two. To obtain CO (ppm) and NMHC (ppm) the readings should be multiplied by 0.05.

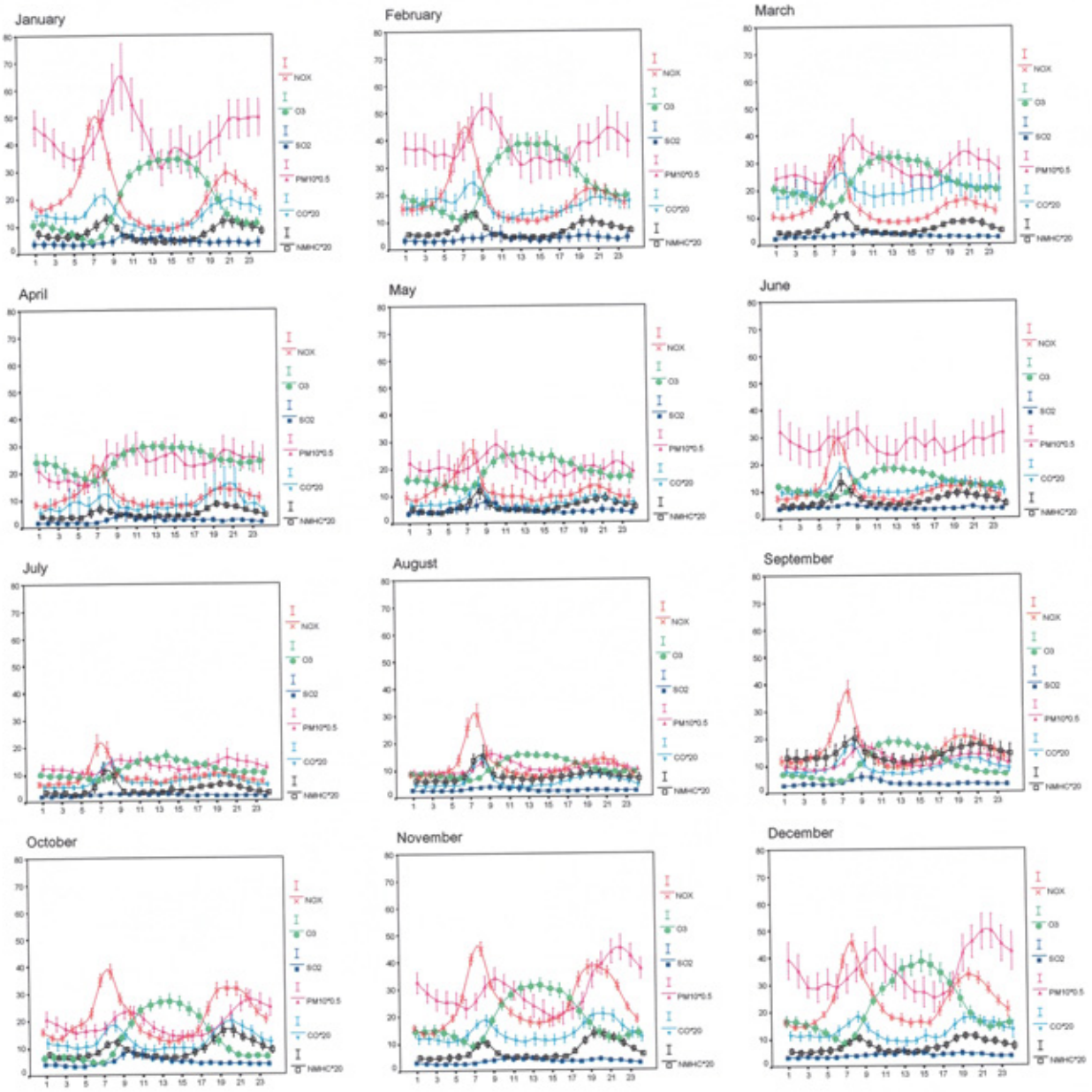


Figure 4.10 Daily variations of pollutant concentrations in each month at 30T station.

Note: Vertical bars represent two standard errors of mean values. The values for NO_x, O_3, and SO_2 are in ppb. To obtain PM_{10} values ($\mu g\ m^{-3}$) the readings from graph should be multiplied by two. To obtain CO (ppm) and NMHC (ppm) the readings should be multiplied by 0.05.

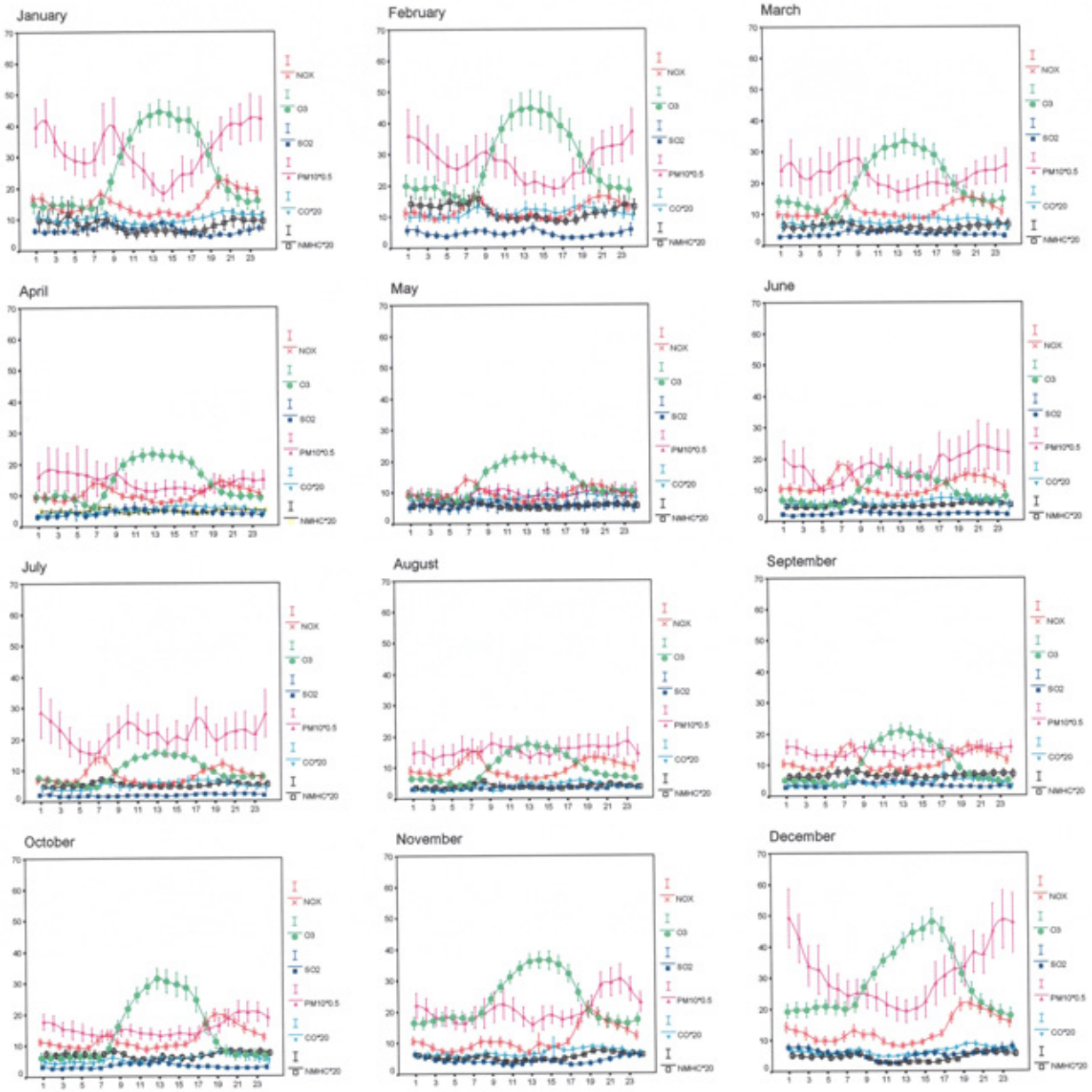


Figure 4.11 Daily variations of pollutant concentrations in each month at 31T station.

Note: Vertical bars represent two standard errors of mean values. The values for NO_x, O_3, and SO_2 are in ppb. To obtain PM_{10} values ($\mu g\ m^{-3}$) the readings from graph should be multiplied by two. To obtain CO (ppm) and NMHC (ppm) the readings should be multiplied by 0.05.

4.4 Traffic Volume

Hourly traffic volume in Passenger Car Equivalent Unit (PCU) was obtained from calculation. The traffic data from Thai Department of Highway, Average Daily traffic (ADT) on Route 3 and Route 36 for road sections near Maptaphut 1997-2001, and the expansion factor from Thai Pollution Control Department were calculated by equation 3.1 and 3.2 in Chapter 3. The hourly volume was presented as shown in Figure 4.12 and 4.13.

Figure 4.12 Diurnal variation of traffic on Route 3 (average 1997 to 2001) (PCU/hour).

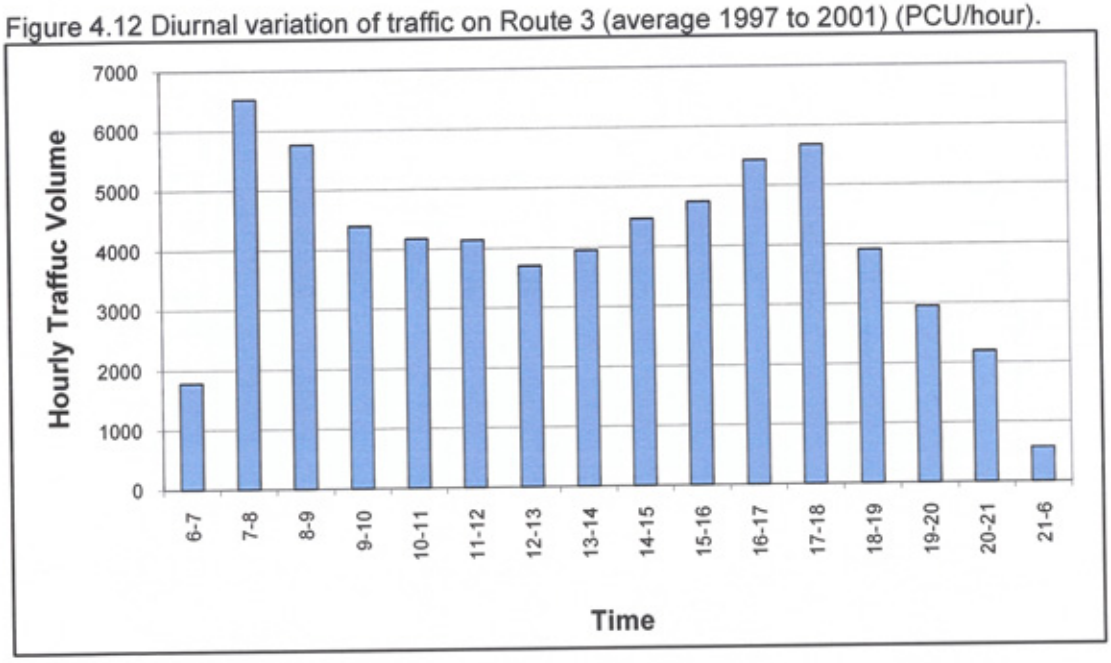
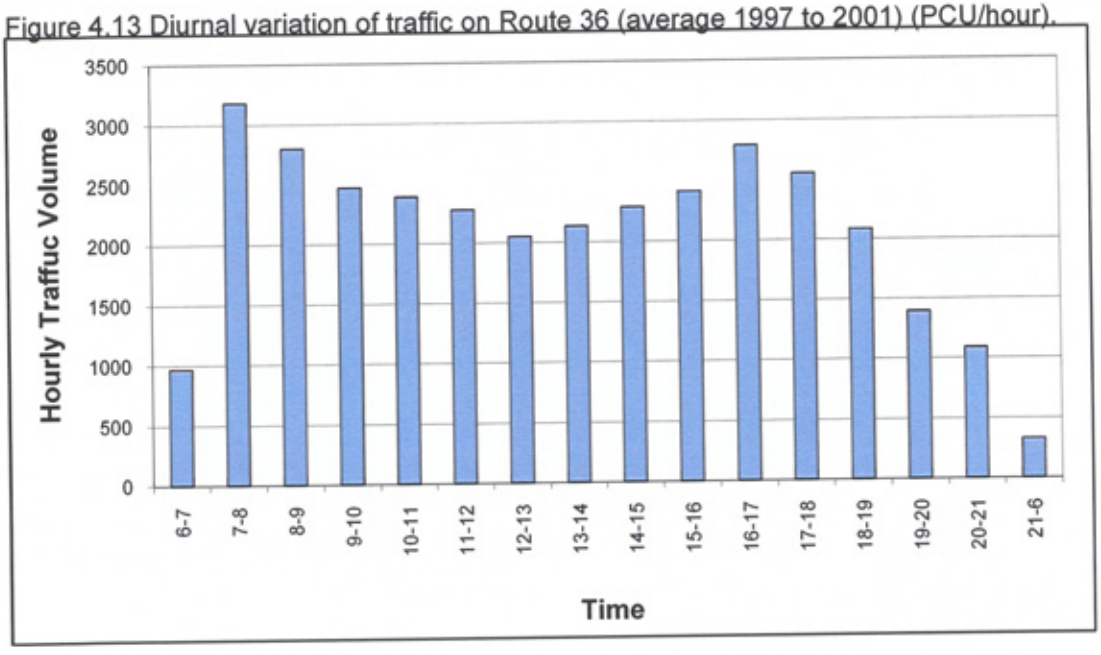


Figure 4.13 Diurnal variation of traffic on Route 36 (average 1997 to 2001) (PCU/hour).



Routes 3 and 36 had the same trend of traffic volume in a day. However, route 3 had twice the density of route 36. The traffic volume was peak at rush hour in the morning (7.00 – 8.00 am.) and in the evening (17.00-18.00 pm.). At night, traffic volume was very low. The lowest traffic volume was during the period 21.00 pm. to 6.00 a.m. on both routes.

4.5 Air Pollution and Health Impact Relationship

The relationship between respiratory sickness data from Maptaphut Hospital and air pollutants data from station 29T during year 2002-2005 was investigated using the multiple linear regression analysis function of the SPSS software package. The numbers of respiratory illness outpatients were collected on a weekly data basis and correlated against all weekly average of air pollutants; the results are shown in Table 4.2. In addition, the effect of cumulative exposure of air pollutant from one week to four weeks was examined (shown in Table 4.2-4.5). It was found that O_3, PM_{10}, and NO are significant to level of more than 95% which indicates that these pollutant gases are correlated with respiratory admissions. It was also shown that cumulative exposure does not significantly change or improve the correlation, with an R^2 value of 41.4% in Table 4.2 for one week exposure and R^2 values of 37.8%, 37.6%, 32.8% for two weeks to four weeks exposure respectively. (Tables 4.3-4.5). This may suggest that a chronic effect is not present in this case. Nevertheless, we cannot definitively conclude that the correlations are as a result of acute effects of PM_{10}, O_3 and NO because the data were based on weekly averages, not daily averages. The Influence of air temperature and %RH were included in the equation to check if they have impacts on rate of respiratory illness admissions as shown in Tables 4.6-4.9. It was found that R^2 increases from 41.4% to 51.2% when temperature and %RH are included in the correlation. Also, accumulation effects of pollutants gases, temperature, and %RH have shown no improvement on correlations.

Weekly data from January 2003 to December 2005 were used. Unfortunately it was not possible to obtain daily respiratory admission data from outpatients in Maptaphut hospitals, nor detail of patients such as age, sex etc. Air pollution data from station 29T were selected as it located near to a residential area and close to the industrial zone. Respiratory outpatient's admission data were only taken from the Maptaphut hospital. Whilst the neighbouring Rayong hospital, some 30 km distant, could have been be included the patients were likely to have been exposed to a variety of different sources, making correlations with Maptaphut emissions difficult. Nevertheless, from a consideration of the admission trends for the two hospitals, as shown in Figure 4.1.4 it seems that the

same factors affecting admissions in Rayong are also prevalent in Maptaphut, perhaps suggesting that emissions from the industrial estate are not a significant factor (since they have little influence on air quality in Rayong).

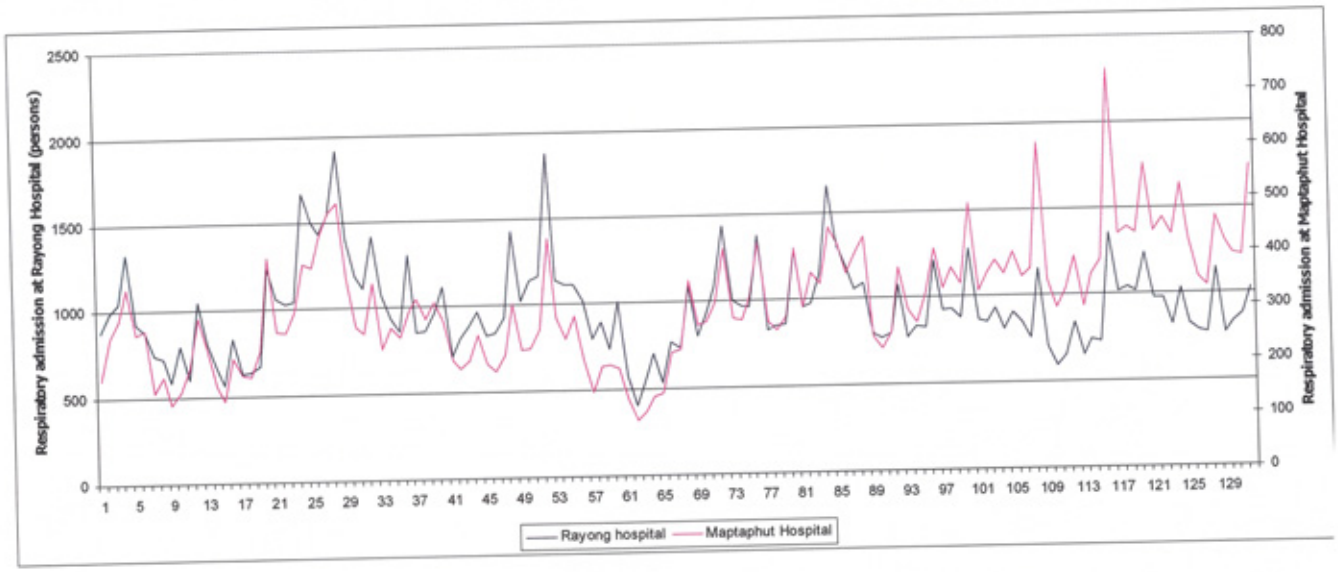


Figure 4.14 Weekly respiratory disease patients at Rayong Hospital and Maptaphut Hospital (January 2003-December 2005).

Multiple linear regression techniques were chosen for this study as there were more than two parameters that can have impact on the admission rates of respiratory disease outpatients. The following terms and definitions were used.

Coefficient of determination (R Square) is a statistic that will give some information about the goodness of fit of a model. In regression, the R^2 coefficient of determination is a statistical measure of how well the regression line approximates the real data points. An R^2 of 1.0 indicates that the regression line perfectly fits the data.

Coefficient of multiple correlation (R) is a quantitative index of association between criterion variable (Y) and set of predictor variables. The square root of the coefficient of multiple determination is the coefficient of **multiple correlation**, *R*.

Standardized regression coefficients (Beta) in any one regression equation are measured on the same scale, with a mean of zero and a standard deviation of 1. They are then directly comparable to one another, with the largest coefficient indicating which independent variable has the greatest influence on the dependent variable.

Unstandardised regression coefficients (B) in any one regression equation cannot compare the size of the various coefficients if the predictor variables are measured on different scales. It means that we cannot directly tell which predictor variable has the most effect on criterion variable (Y).

F-ratio tests statistically significant of R Square.

The standard error of estimate (SE of estimate) indicates the accuracy of a prediction model. The **smaller** the standard error of estimate, the **better** the prediction. The standard error of estimate may be used to construct prediction intervals for the predicted values

4.6 Summary

The first part of this Chapter describes the results of the characterisation study for meteorology and ambient air quality in the Maptaphut area over the period 1998 to 2007; the diurnal variation of meteorological and pollutant concentrations is also described. Generally the pollutant concentrations are below the ambient air quality standards, though ozone concentrations exceeded the 1-hour standard on a small number of occasions in some years (the 8-hour standard was not exceeded on any occasion) and PM_{10} concentrations at all three monitoring stations show regular exceedences of the 24-hour ambient air quality standards.

Hospital admission data for respiratory illness were obtained from both the Maptaphut and Rayong hospitals; the rates from these hospitals show a clear temporal correlation, indicating at first sight that there may be a regional factor that is predominant, with any local factor due to the Maptaphut industrial estate being of secondary importance. However, as will be discussed in more detail in Chapter 5, the smaller Maptaphut hospital is likely to receive a large number of admissions from Rayong City in addition to those from the local community. It is reasonable, therefore, to expect that the two hospital admission profiles will resemble each other since patients may be predominantly from Rayong in both cases.

Notwithstanding the potential confounding factors of uncertainty over admission rates, a significant correlation was found between respiratory admission rates and ozone, PM_{10}

and nitric oxide. Cumulative exposure was not found to improve the correlation, though it was not possible to investigate other lag effects because the data were on a weekly average basis. Lags have been found to be important in literature studies, as discussed in Chapter 2, though the most significant lags are generally of the order of a couple of days at most.

Chapter 5 discusses the results of modelling studies carried out using multiple linear regression (MLR) and the ADMS, ICT3 and PanEIA dispersion modelling packages. The health impacts of pollutants from the industrial estate are further investigated by estimating (from modelling studies) the contribution of the estate to the total ambient pollutant load, and then using the WHO AirQ health impact tool to calculate the likely excess respiratory illness cases based on risk factors derived from the epidemiological studies described in Chapter 2.

Table 4.2 Effect of ambient pollutant gases on respiratory disease admission from Maptaphut hospital on weekly average data during Jan 2003-Dec 2005

Independent Variable	Unstandardised Coefficient		Standardized Coefficient	t	Sig.
	B	SE	Beta		
Constant	31.749	34.011		0.933	0.352
O3	3.446	1.066	0.243	3.232**	0.002
SO2	0.380	2.820	0.012	0.135	0.893
PM10	3.877	0.699	0.432	5.545**	0.000
NO	18.055	4.357	0.366	4.144**	0.000
NO2	2.047	1.839	0.089	1.113	0.268
R	0.643				
R Square (%)	41.4				
SE of estimate	87.289				
F	19.212				
Sig.	0.000				

* p value < 0.05, ** p value < 0.01

Table 4.2 has shown that coefficient of Determination (R square) was 41.4%. Three variables which shown significance level less than 0.05 (Confidence level of 95%) were O_3, PM10 and NO.

Table 4.3 Effect of ambient air pollutant gases on respiratory disease admission from Maptaphut hospital average two weeks exposure during Jan2003-Dec2005

Independent Variable	Unstandardised Coefficient		Standardized Coefficient	t	Sig.
	B	SE	Beta		
Constant	5.996	38.308		0.157	0.876
O3	4.717	1.247	0.310	3.781**	0.000
SO2	-0.534	3.408	-0.015	-0.157	0.876
PM10	4.196	0.933	0.380	4.495**	0.000
NO	26.588	5.286	0.492	5.030**	0.000
NO2	-1.567	2.338	-0.057	-0.670	0.504
R	0.615				
R Square (%)	37.8				
SE of estimate	88.934				
F	16.435				
Sig.	0.000				

* p value < 0.05, ** p value < 0.01

Table 4.3 has shown that coefficient of Determination (R Square) was 37.8%. It has decreased from weekly exposure from R2 41.4% to 37.8%. Three pollutant gases which shown significance less than 0.05 were O_3, PM10 and NO

Table 4.4 Effect of ambient air pollutant gases on respiratory disease hospital average three weeks exposure during Jan 2003 – Dec 2005.

Independent Variable	Unstandardised Coefficient		Standardized Coefficient	t	Sig.
	B	SE	Beta		
Constant	-2.777	40.990		-0.068	0.946
O3	5.911	1.390	0.362	4.252**	0.000
SO2	-2.956	3.958	-0.080	-0.747	0.456
PM10	3.906	1.068	0.316	3.657**	0.000
NO	34.179	6.026	0.598	5.672**	0.000
NO2	-3.895	2.635	-0.127	-1.478	0.142
R	0.613				
R Square (%)	37.6				
SE of estimate	89.202				
F	16.026				
Sig.	0.000				

* p value < 0.05, ** p value < 0.01

Table 4.4 has shown coefficient of Determination (R Square) was 37.6% which decreased from two weeks exposure of R_2 37.8%. Three variable gases were O_3, PM10 and NO providing.

Table 4.5 Effect of ambient air pollutant gases on respiratory disease admission from Maptaphut hospital average four weeks exposure during Jan 2003 – Dec 2005

Independent Variable	Unstandardised Coefficient		Standardized Coefficient	t	Sig.
	B	SE	Beta		
Constant	8.987	44.927		0.200	0.842
O3	6.349	1.574	0.368	4.032**	0.000
SO2	-2.650	4.679	-0.068	-0.566	0.572
PM10	3.339	1.237	0.247	2.699**	0.008
NO	34.234	6.911	0.578	4.953**	0.000
NO2	-4.098	2.946	-0.124	-1.391	0.167
R	0.573				
R Square (%)	32.8				
SE of estimate	92.760				
F	12.789				
Sig.	0.000				

* p value < 0.05, ** p value < 0.01

Table 4.5 shown coefficient of Determination (R^2) was 32.8%. Three pollutant gases which had significant less than 0.05 were O_3, PM10 and NO.

Table 4.6 Effect of pollutant gases including Temperature and %RH on respiratory from Maptaphut hospital on weekly average data during Jan 2003 – Dec 2005.

Independent Variable	Unstandardised Coefficient		Standardized Coefficient	t	Sig.
	B	SE	Beta		
Constant	-411.151	342.400		-1.201	0.234
O3	7.105	2.106	0.401	3.373**	0.001
SO2	1.591	3.861	0.044	0.412	0.681
PM10	4.285	0.900	0.445	4.762**	0.000
NO	13.623	5.796	0.246	2.350*	0.021
NO2	4.621	2.572	0.170	1.797	0.076
TEMP	12.450	10.389	0.112	1.198	0.234
%RH	0.285	1.543	0.020	0.185	0.851
R	0.716				
R Square (%)	51.2				
SE of estimate	87.300				
F	11.399				
Sig.	0.000				

* p value < 0.05, ** p value < 0.01

Table 4.6 shown coefficient of determination (R^2) was 51.2% which increased from 41.4% when Temperature and %RH were not included O_3 and PM10 were having significance of 0.001 and 0.000 respectively.

Table 4.7 Effect of pollutant gases including Temperature and %RH on respiratory disease admission from Maptaphut hospital average two weeks exposure during Jan 2003 – Dec 2005.

Independent Variable	Unstandardised Coefficient		Standardized Coefficient	t	Sig.
	B	SE	Beta		
Constant	-372.182	415.878		-0.895	0.374
O3	8.305	2.729	0.437	3.043**	0.003
SO2	3.620	5.303	0.095	0.683	0.497
PM10	4.583	1.431	0.361	3.203**	0.002
NO	15.876	8.101	0.261	1.960	0.054
NO2	-0.535	3.847	-0.016	-0.139	0.890
TEMP	10.715	12.466	0.098	0.860	0.393
%RH	0.572	1.708	0.041	0.335	0.739
R	0.607				
R Square (%)	36.8				
SE of estimate	98.316				
F	6.249				
Sig.	0.000				

* p value < 0.05, ** p value < 0.01

Table 4.7 shown coefficient of Determination R2 was 36.8% which decreased from weekly average of 57.2%. Two variable gases were O_3 and PM10 providing significant less than 0.05.

Table 4.8 Effect of pollutant gases including Temperature and %RH on respiratory disease admission from Maptaphut hospital average three weeks exposure during Jan 2003 – Dec 2005.

Independent Variable	Unstandardised Coefficient		Standardized Coefficient	t	Sig.
	B	SE	Beta		
Constant	-510.868	413.597		-1.235	0.221
O3	10.915	2.926	0.550	3.731**	0.000
SO2	2.970	6.003	0.077	0.495	0.622
PM10	3.858	1.692	0.269	2.281*	0.025
NO	18.625	9.657	0.288	1.929	0.058
NO2	-1.802	4.731	-0.046	-0.381	0.704
TEMP	11.531	12.223	0.106	0.943	0.349
%RH	1.856	1.821	0.131	1.019	0.311
R	0.592				
R Square (%)	35.0				
SE of estimate	99.350				
F	5.697				
Sig.	0.000				

* p value < 0.05, ** p value < 0.01

Table 4.8 shown coefficient of Determination (R^2) was 35.0%. O_3 Gases was having significance 0.000.

Table 4.9 Effect of pollutant gases including Temperature and %RH on respiratory disease admission from Maptaphut hospital average four weeks exposure during Jan 2003 – Dec 2005.

Independent Variable	Unstandardised Coefficient		Standardized Coefficient	t	Sig.
	B	SE	Beta		
Constant	-538.483	433.438		-1.242	0.218
O3	10.464	3.045	0.514	3.436**	0.001
SO2	3.577	6.571	0.091	0.544	0.588
PM10	3.733	1.896	0.242	1.969	0.053
NO	15.656	11.124	0.234	1.407	0.164
NO2	-0.672	5.586	-0.015	-0.120	0.905
TEMP	13.507	12.689	0.125	1.064	0.291
%RH	1.625	1.868	0.115	0.870	0.387
R	0.561				
R Square (%)	31.5				
SE of estimate	102.011				
F	4.859				
Sig.	0.000				

* p value < 0.05, ** p value < 0.01

Table 4.9 shown coefficient of Determination was 31.5%. O_3 and PM10 providing best significant of 0.001 and 0.053 respectively.

Chapter 5

Results and Discussion: modelling ambient air pollution levels in Maptaphut

Results from Chapter 4 showed that even though pollution levels for the Maptaphut area (ambient monitoring stations 29T and 31T) are generally below the Thai National Ambient Air Quality Standards, there was still a statistical significance observed between hospital admissions for respiratory problems and the ambient concentrations of ozone, nitric oxide and PM_{10}. The latter of these showed the most significant correlation, followed by NO and ozone (for weekly averaging periods). Many recent literature studies have observed health effects at concentrations much lower than the WHO guideline values, so it is not surprising that significant correlations have been observed in this study (see review in WHO 2005).

One of the main objectives of this research was to quantify the effect that industrial emissions from the Maptaphut Industrial estate were having on ambient air quality and, therefore, on the health of the local population. The ambient air quality, as measured at three local monitoring stations has been discussed in the previous chapter; however, it is unclear from this data exactly what the contribution is from the Maptaphut Industrial estate. The use of dispersion modelling techniques offers a well established approach to estimating contributions from pollutant sources, and this study has looked at three different models: ADMS 3, ISC and Panache EIA. The first two of these are modified Guassian models, whilst the other is based on computational fluid dynamics. The objective was to find the most appropriate model for this location since validation studies for all three had tended to used similar data sets, none of which reflected meteorological conditions in this part of Asia. Once validated against the 29T (Maptaphut) ambient monitoring station, the most appropriate model could then be used to estimate the contribution of industrial estate emissions to the ambient concentrations. A model validated against local meteorological conditions is also a vital tool in directing future pollution control policy in the region, allowing policy makers to calculate overall health improvements from a variety of control measures. These studies are discussed in section 5.2.

In addition to dispersion modelling studies, research was carried out into developing statistical models for predicting daily maximum 1-hour PM_{10} and ozone concentrations using meteorological and other pollutant data as parameters in multiple linear regression analysis; this work is discussed in section 5.3.

5.1 Air Pollutant Directional Analysis

As a starting point it was decided to make an assessment of the predominant wind direction associated with each of the main pollutants – in this study this approach is termed this 'air pollutant directional analysis'. It is a form of source apportionment in that it allows the most significant sources for each of the main pollutants to be inferred, be they industrial, road or domestic in their origin. There are many approaches to source apportionment and these often rely on use of dispersion models such as ADMS-Urban, which has the capability of modelling grid, road and industrial sources together. One such study, carried out in London, used ADMS-Urban in combination with an emission inventory to determine that the respective NOx contributions to its area were: 40% major roads, 40% gas burning (domestic) and 20% minor roads (Leksmono *et al.*, 2006). Other source apportionment methods have also been employed, such as the approach of Muir and co-workers who analysed ratios of PM_{10} to other traffic derived pollutants as a means of differentiating between traffic and construction as sources during PM10 pollution episodes; comparisons were also made between hourly PM_{10} concentrations and long term averages, again as a means of identifying construction related pollution (Muir *et al.,* 2006).

The method presented here gives an intuitive rather than a qualitative analysis of the significance of various sources; it provides visual representation against which the success or otherwise of the modelling studies can be analysed. Rudimentary non-normalised directional analyses have been used in published studies. A good example is that of Owen *et al* (1999) where anomalously high sulphur dioxide concentrations obtained for ADMS dispersion modelling studies in London were attributed to power stations in the Thames corridor following a simple analysis of concentration against wind direction.

This technique breaks the wind direction, as measured at the ambient monitoring stations, into 10° segments and analyses the frequency of different pollutant concentration bands for each of the 36 segments. As an example, the combined 2002-2005 data set for NOx at station 29T shows that at a wind direction of 200°, the pollutant concentration lies between 0 and 10ppm for 45% of the time, whereas at 30° the ambient concentration lies within this range for over 98% of the time, indicating a significant difference that is almost certainly due to one of the industrial sources. The location of the main pollutant sources with respect to the monitoring stations are shown in Figure 5.1, with the distances and directions summarised in Table 5.1. Data on the main pollutant sources from the industrial

estate was established from an analysis of some preliminary ADMS modelling studies (discussed later).

The air pollutant directional analysis data is presented as cumulative frequency analysis charts using the Excel 'radar' chart type to depict the wind directions, as shown in Figure 5.2 for each of the main pollutants at each of the monitoring stations. In addition, Figure 5.2e shows the same approach applied to the NO_x/NO_2 ratio, which can be taken as a measure of whether NO_x is coming principally from a road source. Because the frequency analysis normalises all directions to 100% total, the predominance of particular wind directions, for example southerly in the wet season, is negated; this allows us to see the likely direction of pollutant sources with respect to the ambient monitoring stations. This analysis was carried out for three monitoring stations: 29T and 31T in Maptaphut, and 30T in Rayong city; the pollutant sources are likely to be significantly different for these two conurbations.

In analysing the charts presented in Figure 5.2, the predominance of darker colours in one particular direction is an indication of a contributory source. Conversely, a fairly even distribution of darker colours is indicative of the absence of a significant source contributing towards the ambient concentrations for that particular pollutant at that particular point. It should be pointed out that northerly wind directions were much less prevalent than for some other directions, and this has sometimes resulted in 'spikes' or anomalies that are down to low sample sizes rather than significant features.

Examining sulphur dioxide concentrations first (Figure 5.2a), it is clear that for monitoring station 29T (nearest to the industrial estate) there is a 'ripple' of higher concentrations corresponding to the south and south east direction from the source; this is consistent with the main SO_2 emission sources indicated on Figure 5.1, though indicating that the more distant source,SO2-1, is perhaps the main contributor. For ambient station 31T, which is still near Maptaphut, though considerably more distant from the SO_2 sources compared to 29T, there is a much more even distribution of concentrations, though there is evidence of a slight elevation to the south and east, again consistent with the industrial sources, but understandably lower given the greater distance between source and receptor. For this station there is also a small 'ripple' to the north, and there is uncertainty as to what this source might be. Finally for station 30T, in Rayong city, and some 12km from the industrial sources, there is clearly a 'ripple' of higher concentrations, though this is in the opposite

direction to the known SO_2 sources, i.e. to the east. Given that 30T lies to the west of Rayong city, it is most likely that the main contribution to SO_2 or this station is from city-based sources, i.e. from fossil fuel burning, perhaps from generators, cooking and heating systems.

Overall the sulphur dioxide results illustrate the usefulness of this technique in identifying the main contributory sources to ambient air quality at a particular location. Moreover, it demonstrates that this technique can be used in advance to predict the likely effectiveness of dispersion modelling techniques in adequately forecasting ambient pollutant concentrations; if directional analysis results indicate significant sources other than those accounted for in the proposed dispersion model then the forecasting ability of the model is likely to be impaired. The directional analysis technique can, therefore, be used to refine dispersion models, helping to locate and account for other significant sources.

For NO_x, Figure 5.2b indicates that the contributions are fairly evenly distributed at station 29T, with no evidence that the source NO2-1 on Figure 5.1 as a significant contributor to ambient levels. There is, perhaps, some evidence of elevated levels from the east, but not to the south east, where NO2-1 lies. The NO_x/NO_2 ratio analysis, Figure 5.2c, is more revealing, showing the highest ratios as coming from a north westerly direction; higher NO_x/NO_2 ratios are indicative of close-by road emissions where nitric oxide has only partially oxidised to NO_2. The direction in this case is consistent with Highway 3. For 31T, which is much closer to Highway 3, the overall NO_2 levels are fairly evenly distributed, and at similar levels to 29T, but it is also very revealing that the NO_x/NO_2 ratios for this station are significantly higher than for 29T, as would be expected for the road source being the contributory factor. For 30T the NO_2 and NO_x/NO_2 ratios are consistent with the main contributory source being Highway 3.

For the PM_{10} analysis, Figure 5.2d, the main sources appear to be fairly evenly distributed for 29T, though perhaps an indication of slightly elevated levels coming from the north and north-east, most probably from Highway 3. For 31T, the PM10 levels are much higher than at 29T, and the predominant direction of significant contributory sources is to the north indicating other sources other than from Highway 3 to the south. For 30T, it is notable that the PM10 concentrations are much higher than at the other stations with a fairly uniform directional distribution, indicative of a variety of city sources; there are elevations at directions consistent with Highway 3.

Finally, for ozone, Figure 5.2e, which is expected to be the least directional of the pollutants, there is evidence of a southerly bias in all cases, particularly for 30T in Rayong city, perhaps evidence of long range transport from countries to the south, or relatively cleaner air masses drawn down from the north.

On the basis of these results it should be expected that dispersion modelling studies of Maptaphut will perform well for SO_2, however the presence of significant sources for other pollutants means that it is likely that these will be modelled less well. For nitrogen oxides, the model could be expanded to include road sources.

Figure 5.1 Map showing the spatial relationship between ambient monitoring stations, main industrial pollutant sources and roads (base map obtained from Google Earth)

	Ambient monitoring stations					
	T29		T31		T30	
Source	*Distance (km)*	*Angle*	*Distance (km)*	*Angle*	*Distance (km)*	*Angle*
NO2-1	4.8	225	6.4	180	13.9	270
SO2-1	3.0	200	6.4	160	11.7	270
SO2-2	1.3	250	4.2	150	12.1	280
PM-1	3.7	210	6.7	165	12.3	270
PM-2	2.9	240	4.8	170	13.1	275
PM-3	1.6	260	3.2	140	12.5	285
PM-4	5.9	300	5.1	210	16.4	275
Road 3	0.76		0.37		0.39	

Table 5.1 Summary of distances and angles from the ambient monitoring points to the main pollutant sources indicated on Figure 5.1

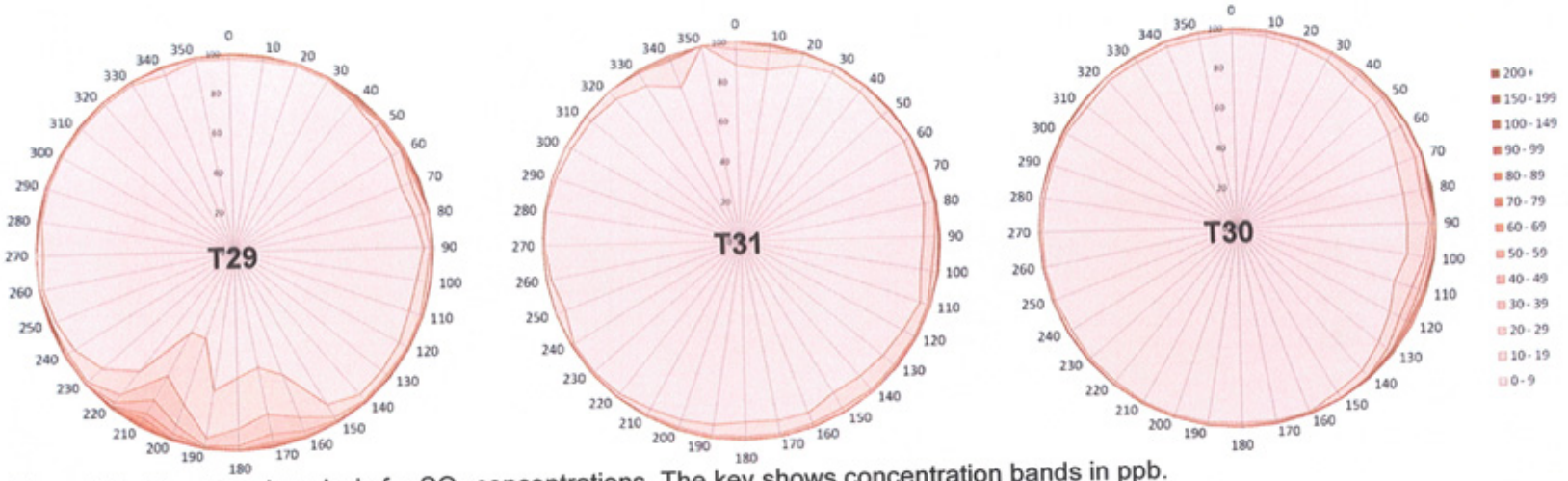


Figure 5.2a Directional analysis for SO_2 concentrations. The key shows concentration bands in ppb.

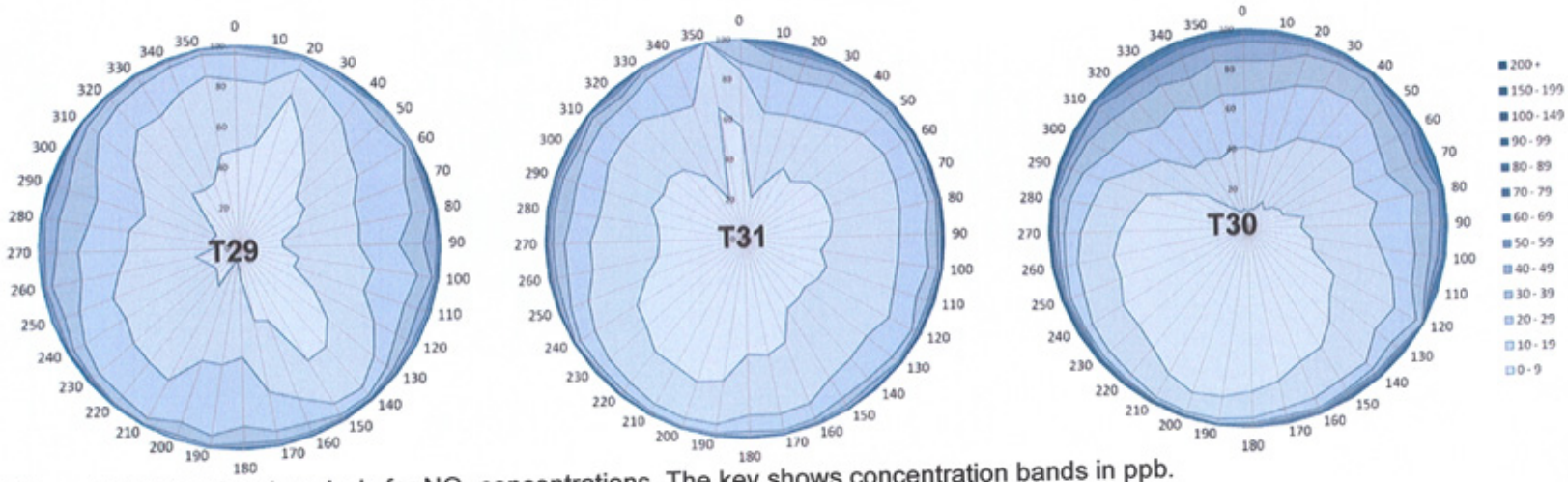


Figure 5.2b Directional analysis for NO_x concentrations. The key shows concentration bands in ppb.

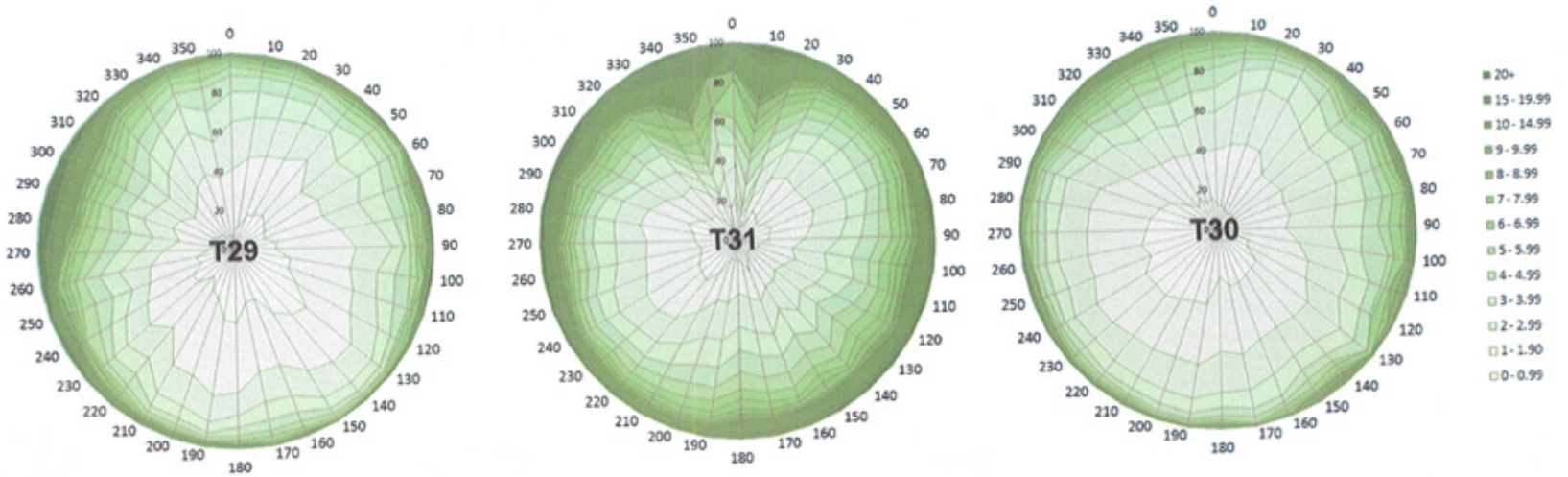


Figure 5.2c Directional analysis for NO_x/NO_2 ratios. The key shows ratio bands.

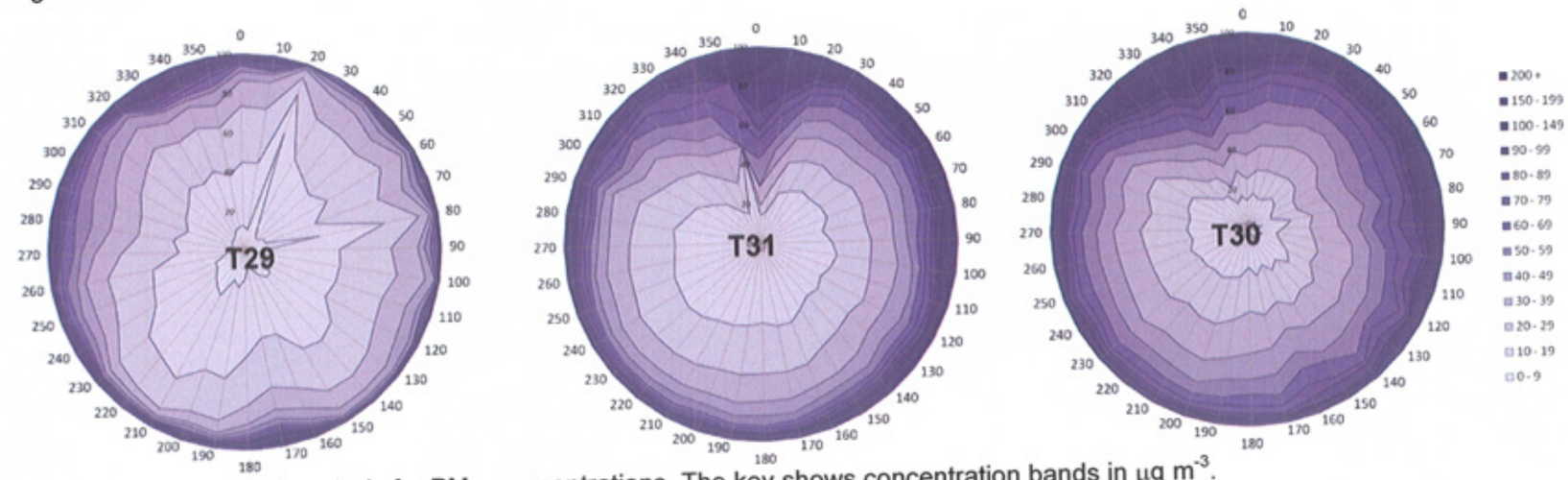


Figure 5.2d Directional analysis for PM_{10} concentrations. The key shows concentration bands in $\mu g\ m^{-3}$.

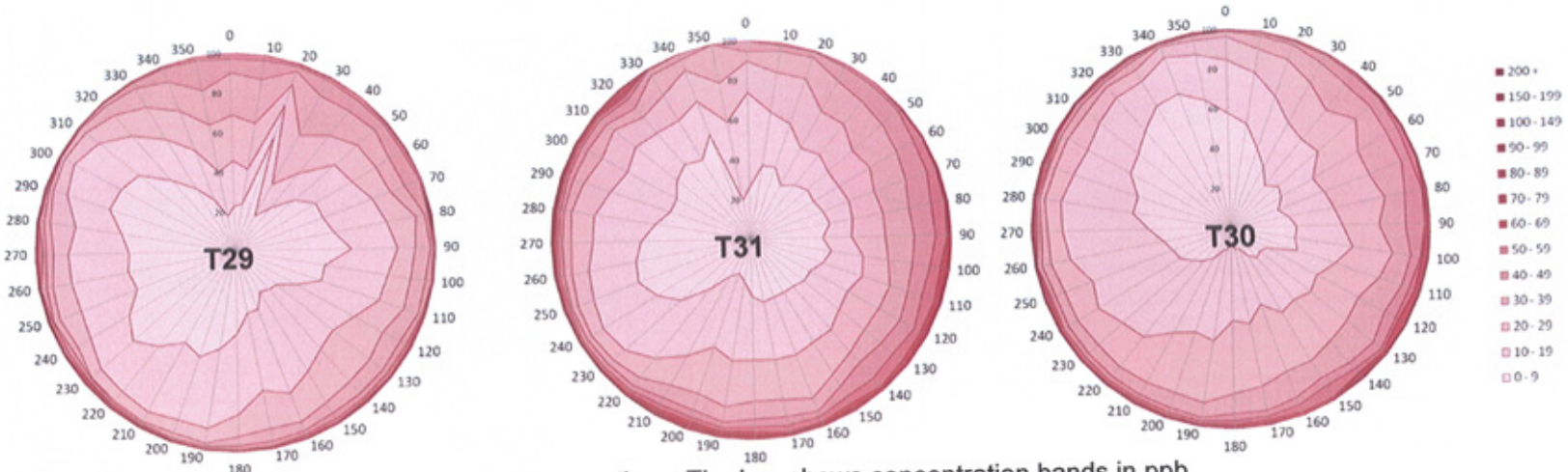


Figure 5.2e Directional analysis for ozone concentrations. The key shows concentration bands in ppb.

5.2 Dispersion modelling results

Dispersion modelling studies were carried out for the location of 29T ambient air quality monitoring using three models: ADMS, ISC and Fluidyn PanEIA; the first two of which are modified Gaussian models, and the latter based on computational fluid dynamics.

Figures 5.3, 5.4 and 5.5 show time series comparisons of observed and ADMS-predicted concentrations for SO_2, NO_x and PM_{10} for the years 2003, 2004 and 2005 respectively; similar comparisons are shown in Figures 5.5 to 5.8 for ISC predicted concentrations and in Figure 5.12 for PanEIA modelled results, though in the latter case modelling was carried out only for 2005. Data for the time series comparisons is presented mainly as 5-day rolling averages for clarity of comparison, though for ADMS modelled SO_2 in 2003 (Figure 5.3a) daily averages are also presented, as an example. For the PanEIA data (Figure 12) it was not possible to calculate 5 day rolling averages due to the interrupted predicted data set and so these data are presented as daily averages.

As well as time series comparisons, correlations have been made between observed and predicted *daily* average data over three year periods for the ADMS (2003 to 2005) and ISC (2002 to 2004); this analysis is presented in Figures 5.9, 5.10 and 5.11 for SO_2, PM_{10} and NO_x respectively. For the PanEIA data, a correlation is presented for SO_2 only (Figure 5.13)

5.2.1 General overview of results

Taken as a whole, the results are fairly mixed, though it is clear that ADMS 3 has by far the superior predictive power, particularly for SO_2. The performance of ISC was surprisingly poor for all pollutants, whereas PanEIA predicted the general trends correctly for SO_2 but had a tendency to over-predict by a factor of two for this pollutant; the predicative capability of PanEIA for PM_{10} and NO_x was generally poor.

5.2.2 Discussion of ADMS results

ADMS has been shown to predict sulphur dioxide concentrations with a reasonable degree of accuracy, as shown in Figures 5.3a, 5.4a and 5.5a for the years 2003, 2004 and 2005 respectively; both the magnitude of the peaks and the general trends are reasonably characterised as evidenced by the observed vs. predicted correlation shown in Figure 5.9a. In many respects, for this particular study, the modelling of sulphur dioxide is the best test for the dispersion modelling programs since the only significant source is the industrial estate; for both PM_{10} and NO_x there are significant other sources, as indicated by the directional analysis studies discussed in Section 5.1. Clearly then, the relatively successful prediction of sulphur dioxide concentrations at this monitoring point indicates that the meteorology model used in ADMS, and the treatment of the boundary layer and Monin-Obukhov lengths is applicable to South East Asian climatic conditions. It is also noteworthy that these correlations were obtained without the use of many of the advanced ADMS features, i.e. chemistry, precipitation, terrain and coastline.

Literature studies of the performance of ADMS to predict SO_2 concentrations, whilst often producing good agreement with measured values, have nevertheless highlighted a number of problems. For example a UK study involving three very different urban environments (London, Belfast and Neath/Port Talbot) showed that SO_2 predictions were affected by (a) unaccounted industrial emissions (power stations in the Thames estuary), (b) complex terrain (Belfast and Port Talbot / Neath: predictions improved once this was taken into account) and (c) the steady state nature of the model, i.e. the assumption that meteorological conditions at the source will prevail for sufficient time for the pollutant to travel to the receptor (Carruthers *et al.*, 2000).

For NO_x predictions, clearly there is a background element, as would be expected from an urban/industrial zone (Figures 5.3c, 5.4b and 5.5b). In the absence of a measured background value, we have estimated from a comparison of the modelled and monitored data that this is approximately 12 $\mu g\ m^{-3}$; background corrected time series plots are shown in Figures 5.3d, 5.4c and 5.5c for the three study years. The prediction trends are rather contradictory, with over-prediction occurring during the period of July to September (rainy season, southerly wind direction with low wind speeds) and an under prediction in the other periods. The obvious conclusion would be that wet deposition needs to be included in the model; however this was tried in the present study and made very little difference to the results (not shown). It is possible that that the discrepancies can be partly

attributed to the absence of the chemistry module during the runs; essentially we are assuming that the emissions from the stacks remain unchanged during their travel from stack to receptor (ADMS default of 10% NO_x as NO_2 was used). Nevertheless a consideration of literature studies shows that NO_X and NO_2 prediction does appear to be variable using the ADMS family of programs. In one study, comparing statistical techniques against ADMS-Urban for a street canyon, the latter technique was found to under predict NOx levels by 11% and NO2 levels by 20% (Westmoreland *et al.*, 2007). Similar under-predictions were seen in a comparison of ADMS-Roads against a Lagrangian model (LASAT) for street canyon modelling in Berlin, Stockholm and London (Hirtl & Baumann-Stanzer, 2007). For urban areas, in a study of NO_x and SO_2 predictions in London the NO_x concentrations were only slightly overestimated in summer, though during winter significant underestimation occurred, most probably due to the lack of seasonality in the emissions inventory (Owen *et al.*, 1999); it is possible that this reason may also be valid for the Maptaphut study – constant emission rates were assumed throughout the year, and this may not necessarily be the case for the principal NO_x emissions – further investigations would need to be carried out. Low wind speeds have also been suggested as reasons for under-predictions (Carruthers *et al.*, 2000).

Regarding the PM_{10} predictions using ADMS (Figures 5.3e, 5.4d and 5.5d for the three study years), clearly as with the NO_x data, and as indicated as likely from the directional analysis, there is a significant background of between 10 to 20µg m^{-3} that may include a variety of sources including: conversion of SO_2 to sulphate particles, sea salt, construction dust, general windblown dust, traffic pollution and domestic sources. It is evident also, however, that the industrial estate is making some contribution since some of the observed and predicted peaks coincide. Generally PM_{10} is predicted poorly by ADMS, even with the inclusion of a background concentration (graph not shown). The 2004 data are particularly poor, though evident technical problems with the monitoring station, resulting in long periods of downtime, may indicate that this year's data are unreliable. For the other two years, as with the NO_x results, there seems to be an over prediction by ADMS in the June to September period and an under prediction at other times (if the background is added). Wind speed may be a factor in this case, with the 2003 data in particular showing higher background levels at times of higher wind speeds, indicating that general windblown dust represent be a significant contribution. During the rainy season, there may be significant particulate removal from the atmosphere and 'damping down' of dust on the ground. These factors were found to be significant in a study that looked at the applicability of

ADMS for air quality assessment and management in two Chinese cities, Fushun and Jinan. In both cities windblown dust sources made significant contributions to the PM_{10} load, with the main problem being extensive areas of bare ground poorly covered with vegetation, construction work and slag heaps; ADMS was able to successfully model the total suspended particulate concentrations, achieving good agreement between monitored and modelled annual mean and maximum concentrations (McHugh *et al*, 2005). The model took account of road sources, stacks domestic sources and windblown dust; the latter was treated using area sources in which emissions were related to wind speed, with no emissions assumed for speeds below 5m/s. Clearly for our model, the main sources, other than the industrial estate, have been insufficiently characterised and would need further work. The directional analysis work showed that the main contributions were fairly evenly distributed with direction, with perhaps the easterly direction representing the least significant contribution (surprisingly, perhaps, since Rayong lies in that direction).

Interestingly, other studies have also shown that ADMS has a tendency to under-predict PM_{10} concentrations. The Carruthers study referred to in the discussion of NO_x showed that PM_{10} was significantly underestimated for the Neath / Port Talbot area, which was attributed to long rage transport sources not being included (Carruthers *et al.*, 2000). The same study showed similar under-predictions in Belfast; this was attributed to regional variations, incorrect emission factors and conversion of locally emitted SO_2 to sulphate particles. Similar factors may apply to the Maptaphut study. For PM_{10} modelling in areas where the main source is roads, the performance of ADMS has been found to be generally good, for example the results of a study reported for Bologna in Italy found that predicted and measured *monthly average* concentrations lay within +/- 30% of each other (Piersanti et *al.*, 2005).

The use of a wide range of dispersion models for predicting particulate concentrations have been extensively reviewed by (Holmes & Morawska, 2006) the conclusions for ADMS was that it does tend to under-predict though as discussed above, this may often be down to poorly characterised sources.

5.2.3 Results for ISC

The predicative capability of ISC for SO_2, NO_x and PM_{10} is generally very poor, as shown in the time series comparisons in Figures 5.6 to 5.8 and the correlations shown in Figures

5.9b, 5.10b and 5.11b. There is some coincidence of peaks, but generally the peaks are over predicted for SO_2 and NO_2 and under predicted for PM_{10}. Some of the factors contributing to the relatively poor NO_x and PM_{10} predictions when using ADMS may also apply here, though ISC3 incorporates a much simpler model of vertical turbulence within the atmospheric boundary layer compared with ADMS and AEROMOD both of which use a bimodal distribution of vertical turbulence within the atmosphere; clearly this is an important factor.

As discussed in Chapter 2 a number of evaluatory studies have been carried out on ISC. For example, Hanna *et al* (1999) carried out a comprehensive comparative study that looked at the performance of ADMS, AEROMOD and ISC3 in predicting mean and maxima ground level concentrations for five sets of test data representing buoyant and non-buoyant plumes of SF_6 released from sources in representing rural urban and complex terrain. For the non-buoyant releases, ISC tends to over-predict the maximum concentrations by up to a factor of ten, compared to ADMS which under-predicts the maximum concentration by up to 40%. For the buoyant releases only 13% of concentrations predicted by ISC were within a factor of two of the measured concentration, compared to 59% for ADMS (Kincaid test set). Perhaps, then, it is unsurprising that ISC has performed so poorly for this dataset.

5.2.4 Results for PanEIA

Figures 5.12a, 5.12b and 5.12c for SO_2, NO_x and PM_{10} respectively (all 2005) show that whilst this computational fluid dynamics (CFD) dispersion model is able to predict general trends and peaks, particularly for SO2, there is a tendency to significantly overestimate the daily average concentrations. Literature comparisons between CFD models (FLUENT) and ADMS have concluded that they give comparable results but that ADMS takes considerably less run-time and that CFD is only really appropriate in applications involving complex geometry that can't be simulated using ADMS's terrain and building modellers (Riddle *et al.*, 2004). Our experience of PanEIA concurs with that of Riddle, i.e. that run times are significantly greater than those for either ADMS or ISC, possibly by a factor of 10.

There are relatively few other literature studies that have compared the performance of CFD and Gaussian dispersion models, though Holmes and Morawska (2006) reports the

success of the MISKAM CFD model in predicting annual mean concentrations of pollutants. The authors also stress the importance of correct selection of the turbulence parameters; we also found significant variations in ground level pollutant concentrations for different turbulence models. In fact, because the terrain was not modelled, as with the ADMS model, it probably wasn't a fair comparison of the performance of PanEIA with ADMS since the former is specifically designed for complex terrain and building environments.

There are no literature studies that have been published specifically on PanEIA, however the authors of the software have published a validation report based on standard Kincaid (buoyant SF_6 release from stack), Copenhagen (non-buoyant SF_6 release from tower) and Lillistrom (SF_6 release from a mast, over a flat residential area with 6 – 10m high buildings and trees) (Fluidyn 2006). For Kincaid the deviations ranged from 7 – 40% for up to 10km downwind of the stack; for Copenhagen the predicted values deviated from the measured by between 6 and 27% for up to 5.4km downwind; and for Lillestrom the deviations were between 2 and 24%. The report makes comparisons with the performance of other software, for example, the maximum SF_6 concentration from the Copenhagen dataset is predicted by PanEIA within 0.1% yet for ADMS there is a 75% under prediction; this supports the view that CFD models are better able to predict peak concentrations.

5.2.5 Contour plots

In addition to the time series analysis, contour plots are an effective method of presenting the results of dispersion modelling studies. Figures 5.13 to 5.15 show, for example, maximum 1-hour concentrations of SO_2 for the years 2002 to 2004 for both ADMS and ISC. ADMS modelled data lies in the range 250 – 800 µg m^{-3} and ISC in the range 372-1,040 µg m^{-3}; these compare to the hourly maximum standard of 780 µg m^{-3}. Based on the more reliable ADMS data, this maximum value is exceeded only near to the sources themselves.

In addition to the yearly maximum values of 1 hour concentrations, seasonal plots for ISC only are shown in Appendix E.

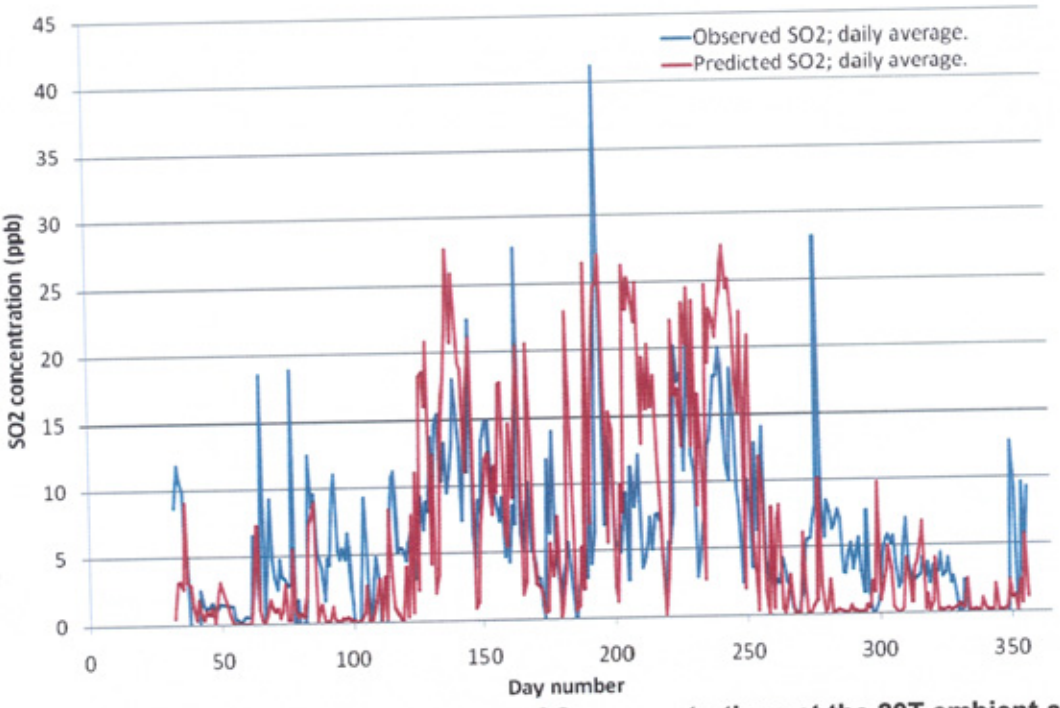


Figure 5.3a Comparison of daily average SO_2 concentrations at the 29T ambient air quality monitoring station for 2003 with those predicted by ADMS.

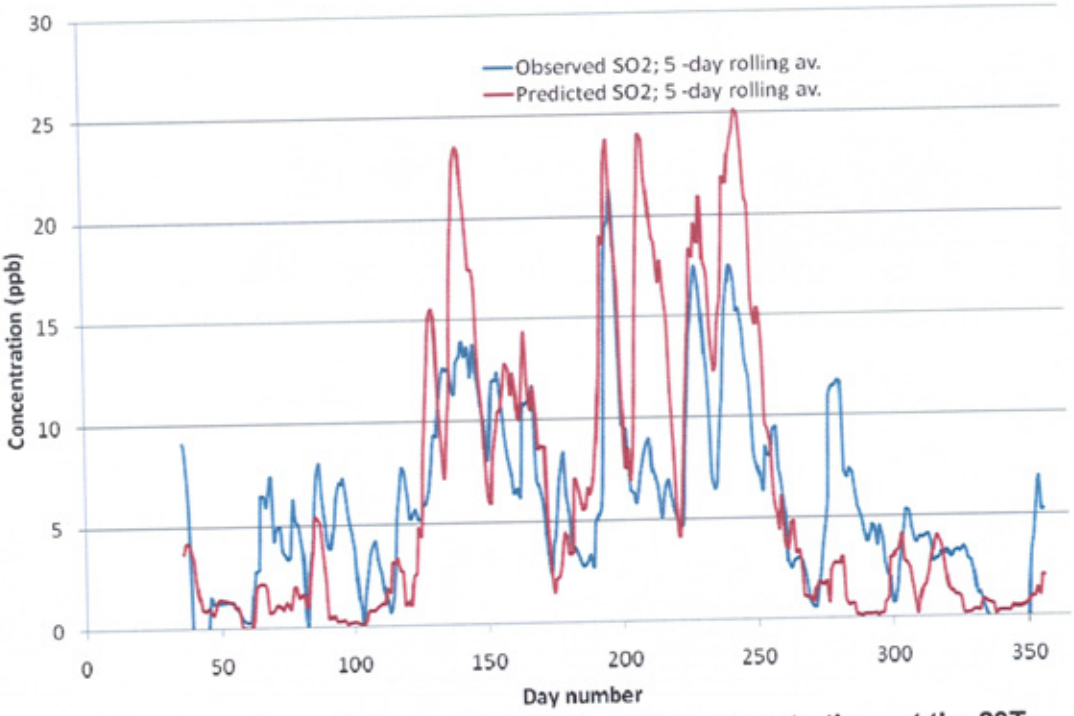


Figure 5.3b Comparison of 5-day rolling average SO_2 concentrations at the 29T ambient air quality monitoring station for 2003 with those predicted by ADMS.

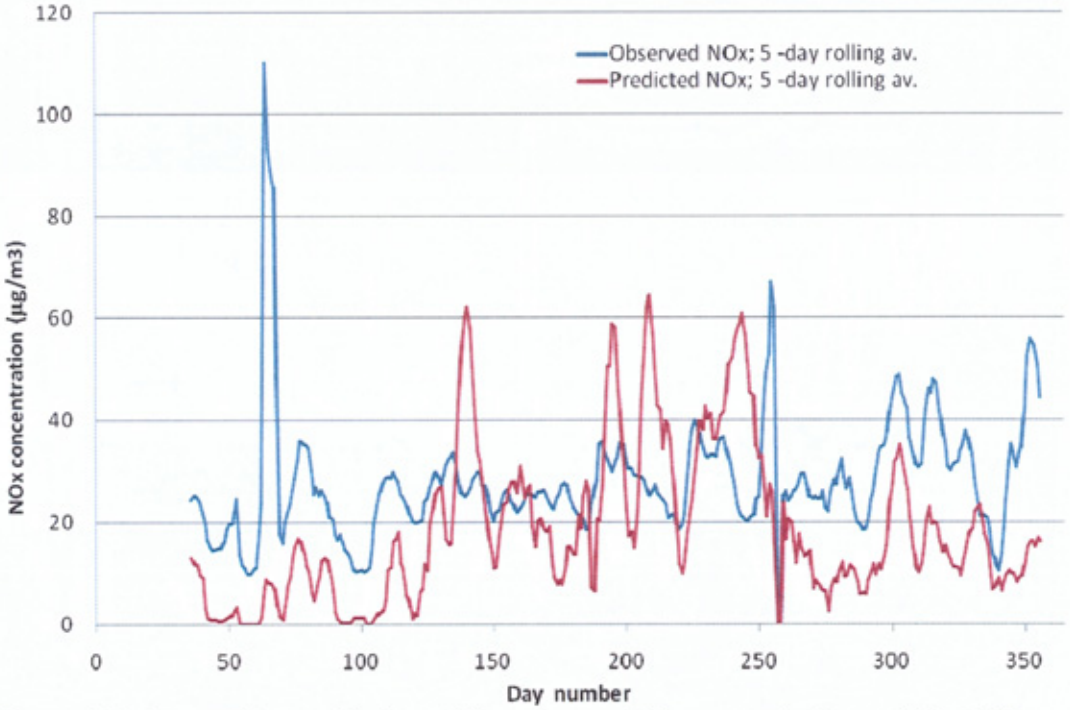


Figure 5.3c Comparison of 5-day rolling average NO_x concentrations at the 29T ambient air quality monitoring station for 2003 with those predicted by ADMS.

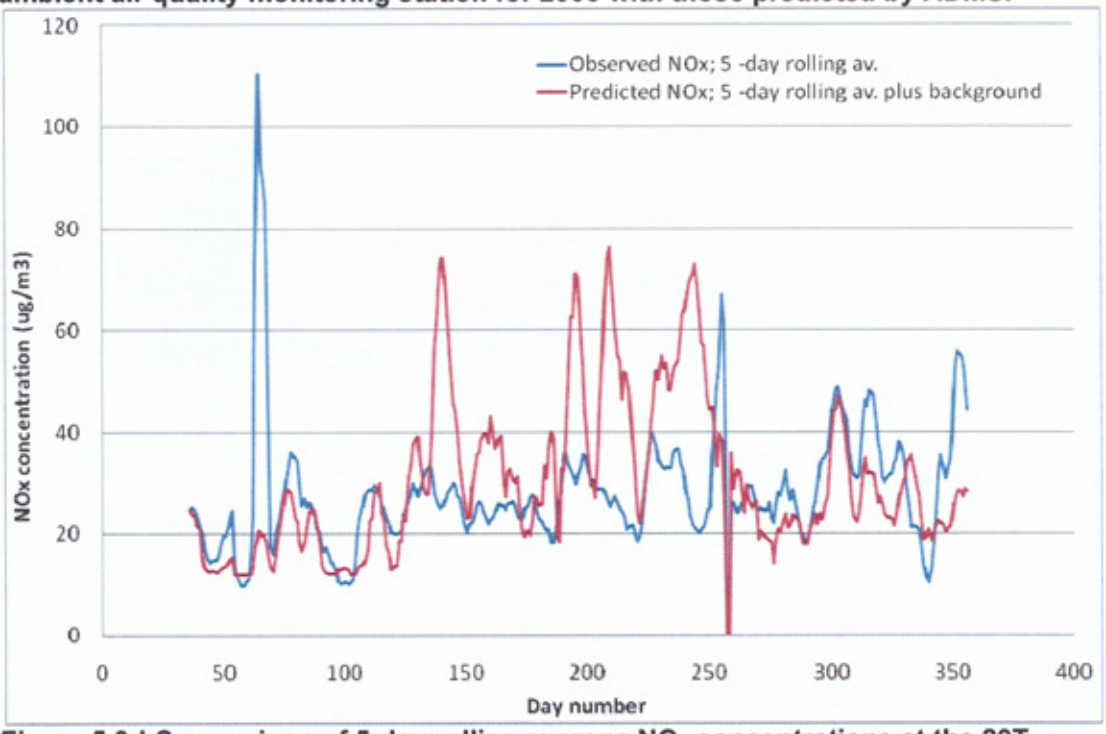


Figure 5.3d Comparison of 5-day rolling average NO_x concentrations at the 29T ambient air quality monitoring station for 2003 with those predicted by ADMS, assuming a 12 µg m^{-3} background concentration.

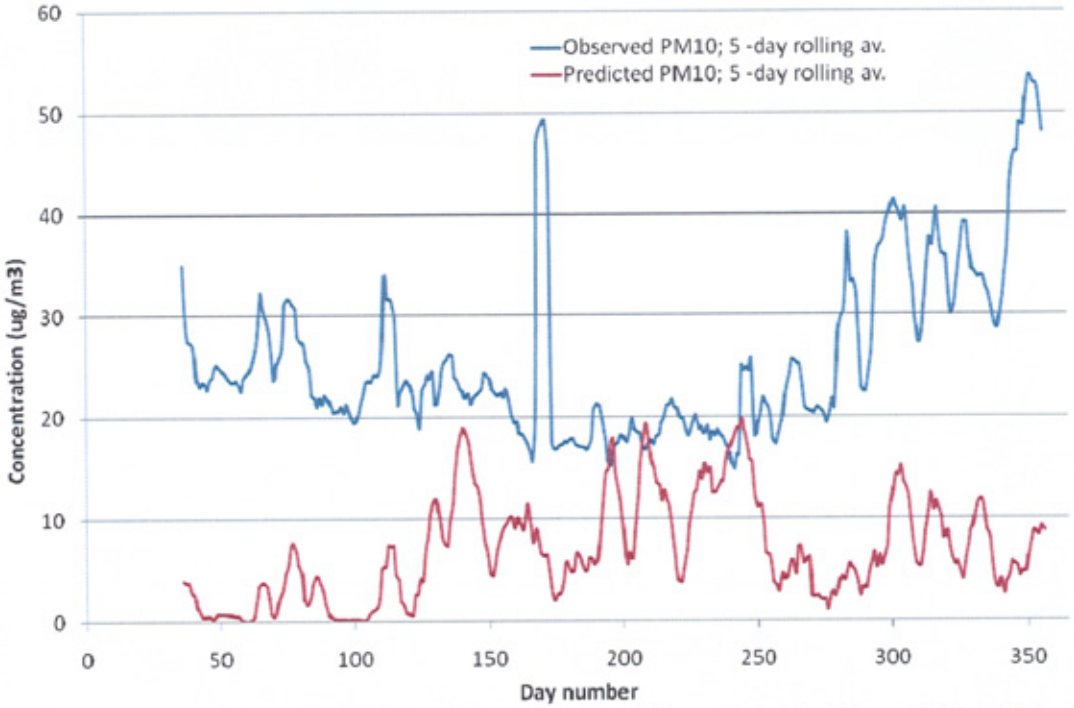


Figure 5.3e Comparison of 5-day rolling average PM_{10} concentrations at the 29T ambient air quality monitoring station for 2003 with those predicted by ADMS.

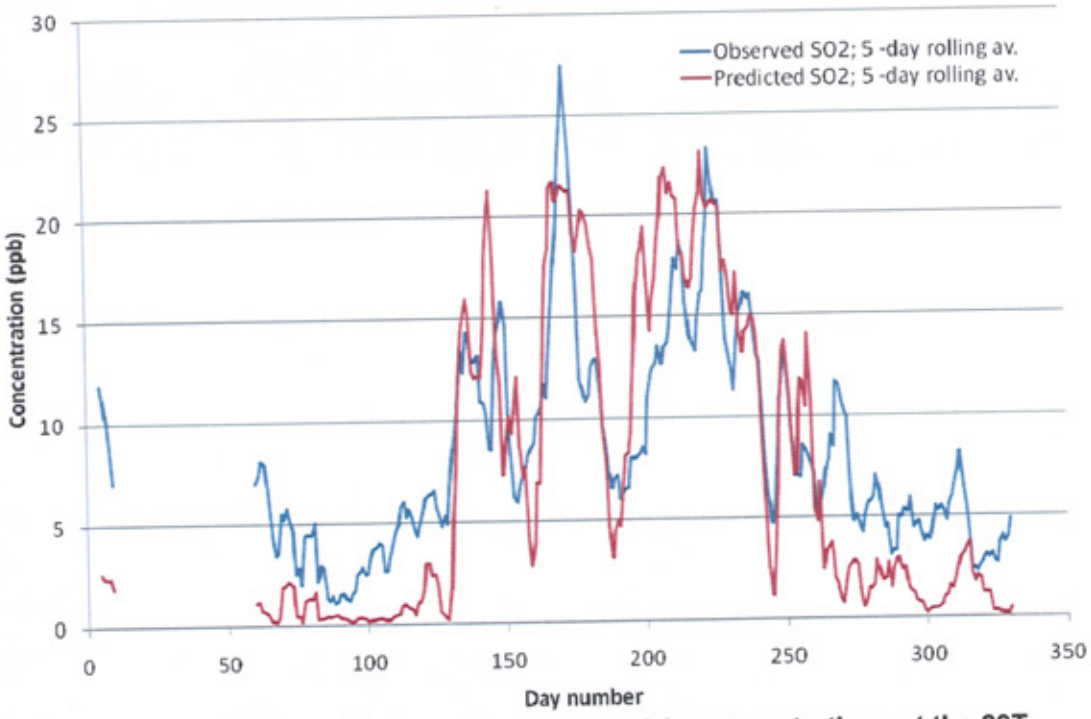


Figure 5.4a Comparison of 5-day rolling average SO_2 concentrations at the 29T ambient air quality monitoring station for 2004 with those predicted by ADMS.

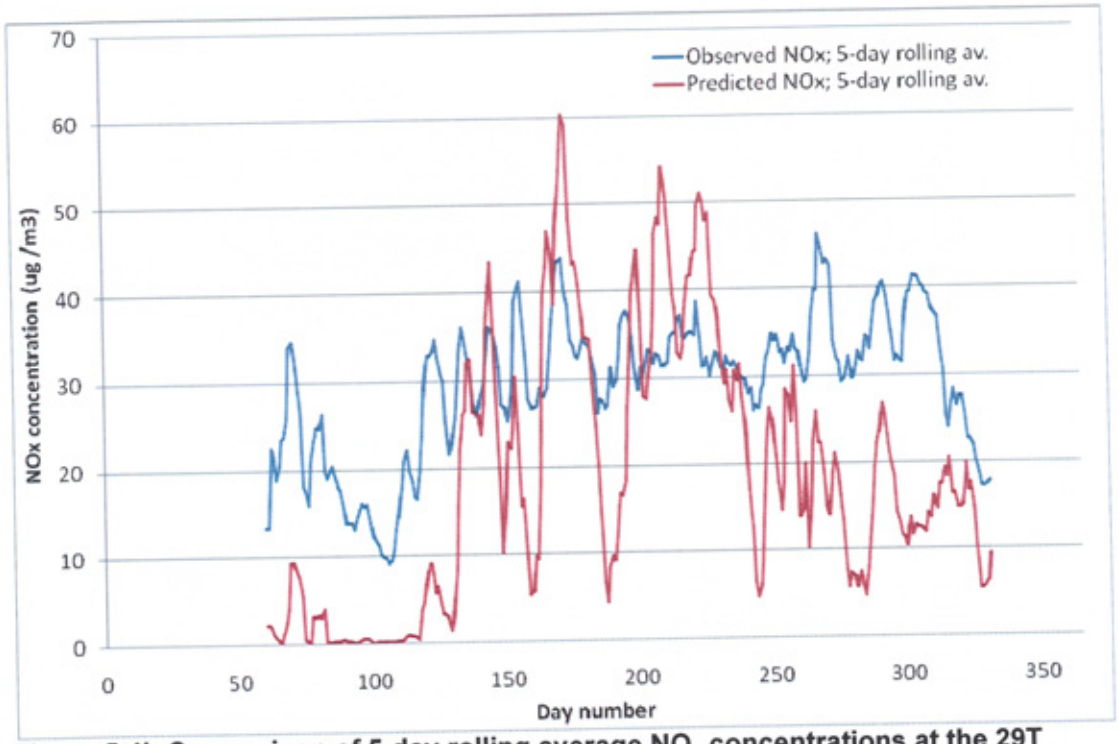


Figure 5.4b Comparison of 5-day rolling average NO_x concentrations at the 29T ambient air quality monitoring station for 2004 with those predicted by ADMS.

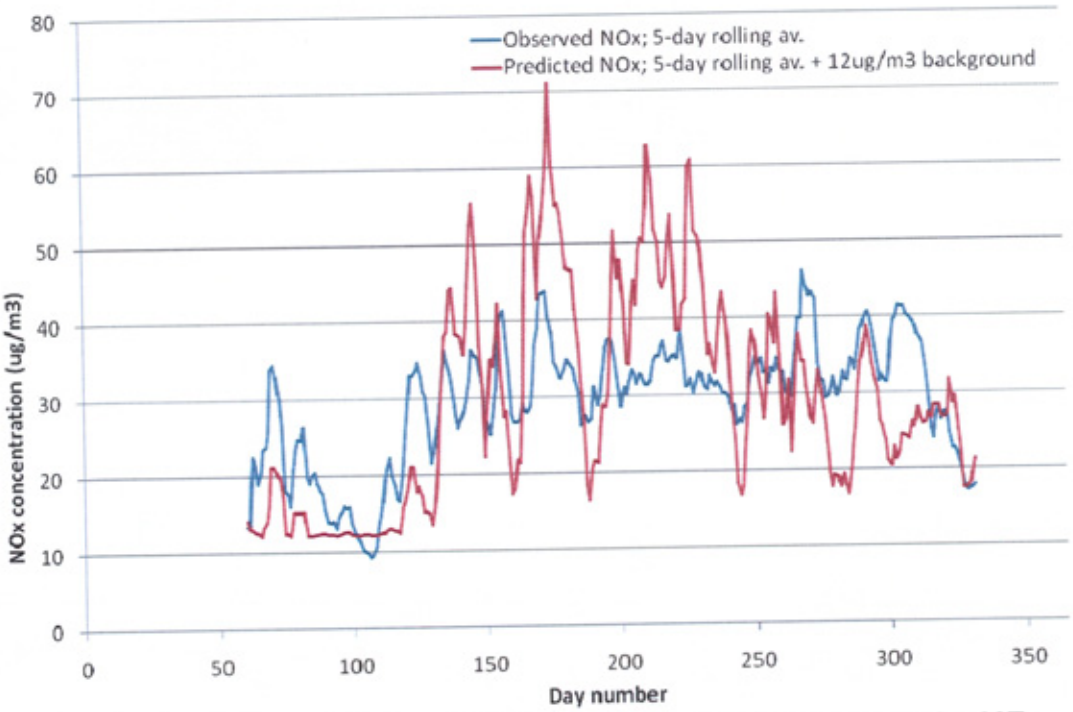


Figure 5.4c Comparison of 5-day rolling average NO_x concentrations at the 29T ambient air quality monitoring station for 2004 with those predicted by ADMS, assuming a 12 μg m^{-3} background concentration.

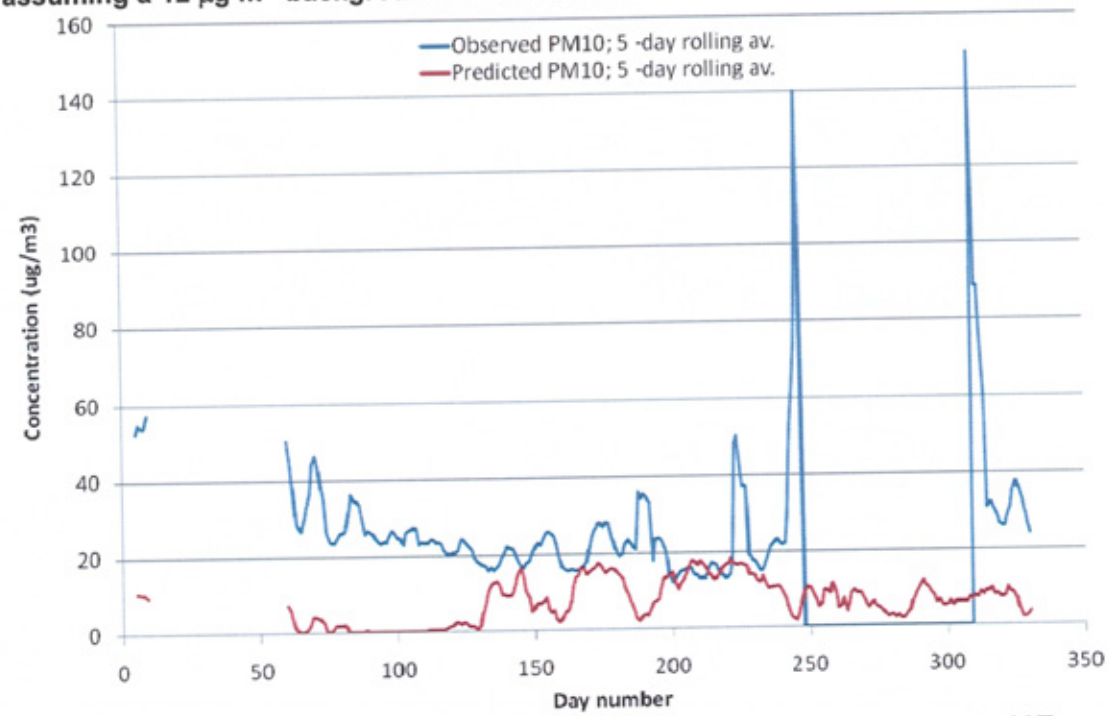


Figure 5.4d Comparison of 5-day rolling average PM_{10} concentrations at the 29T ambient air quality monitoring station for 2004 with those predicted by ADMS.

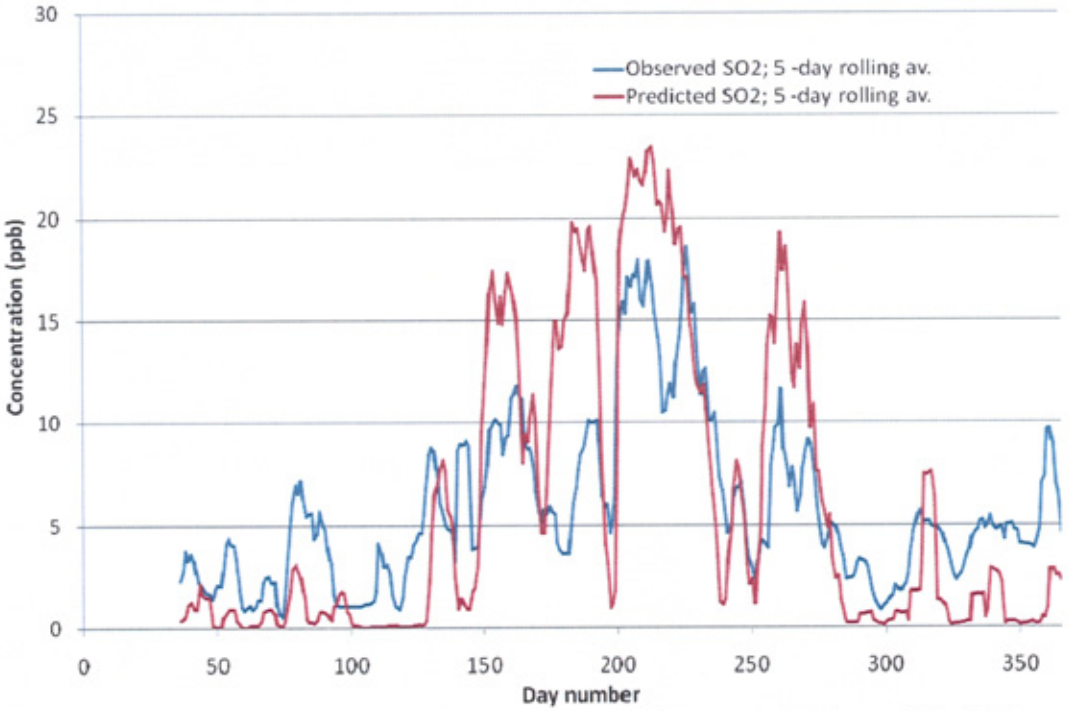


Figure 5.5a Comparison of 5-day rolling average SO_2 concentrations at the 29T ambient air quality monitoring station for 2005 with those predicted by ADMS.

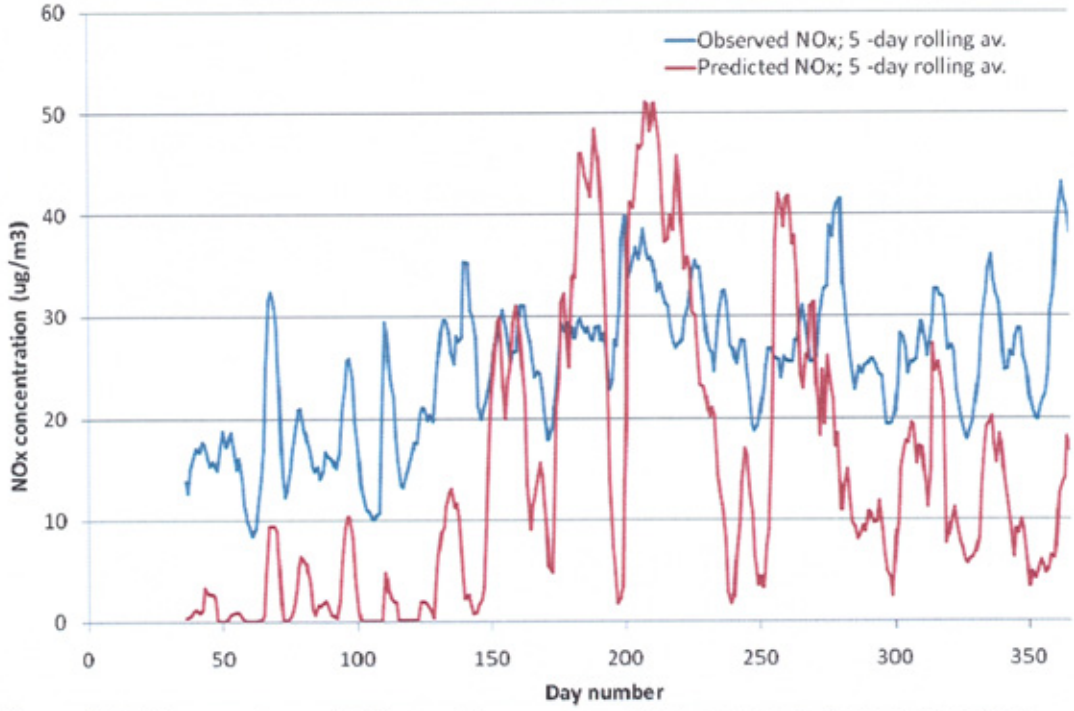


Figure 5.5b Comparison of 5-day rolling average NO_x concentrations at the 29T ambient air quality monitoring station for 2005 with those predicted by ADMS.

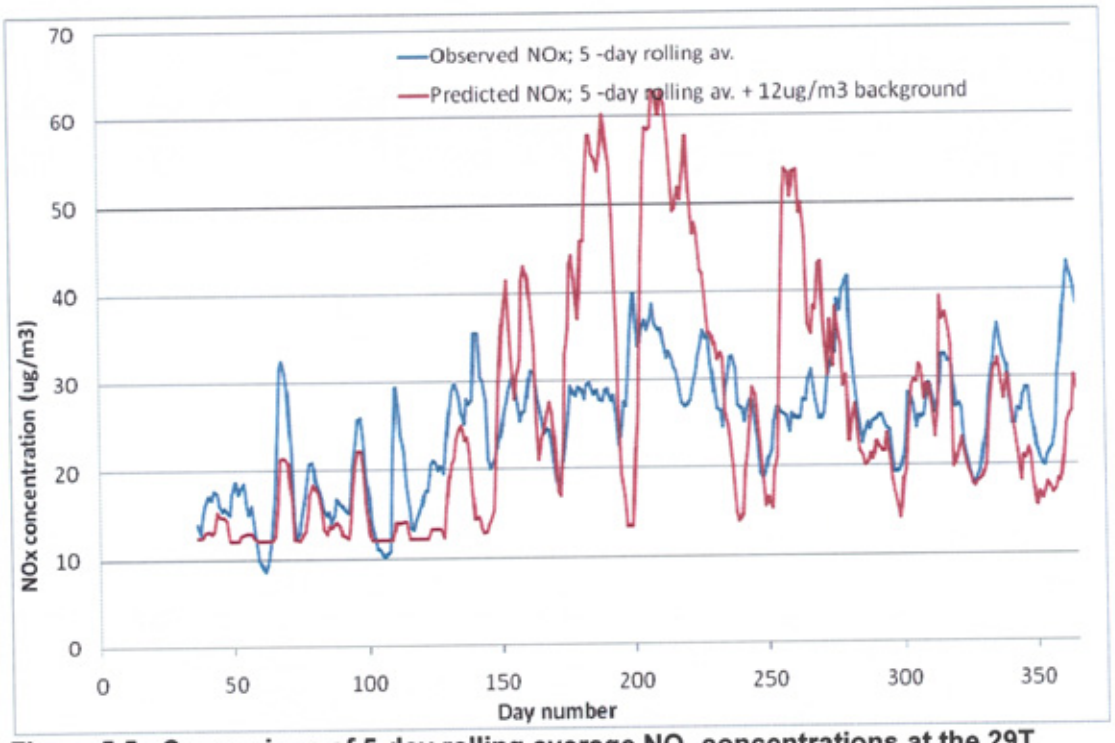


Figure 5.5c Comparison of 5-day rolling average NO_x concentrations at the 29T ambient air quality monitoring station for 2005 with those predicted by ADMS, assuming a background concentration of 12μg m^{-3}.

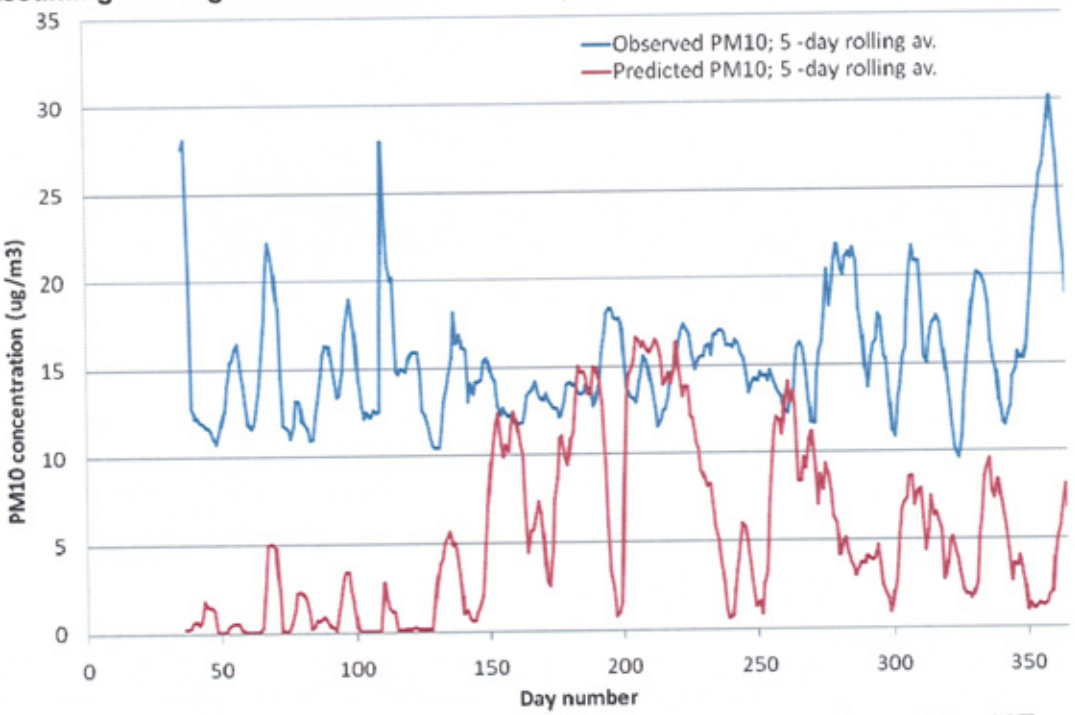


Figure 5.5d Comparison of 5-day rolling average PM_{10} concentrations at the 29T ambient air quality monitoring station for 2005 with those predicted by ADMS.

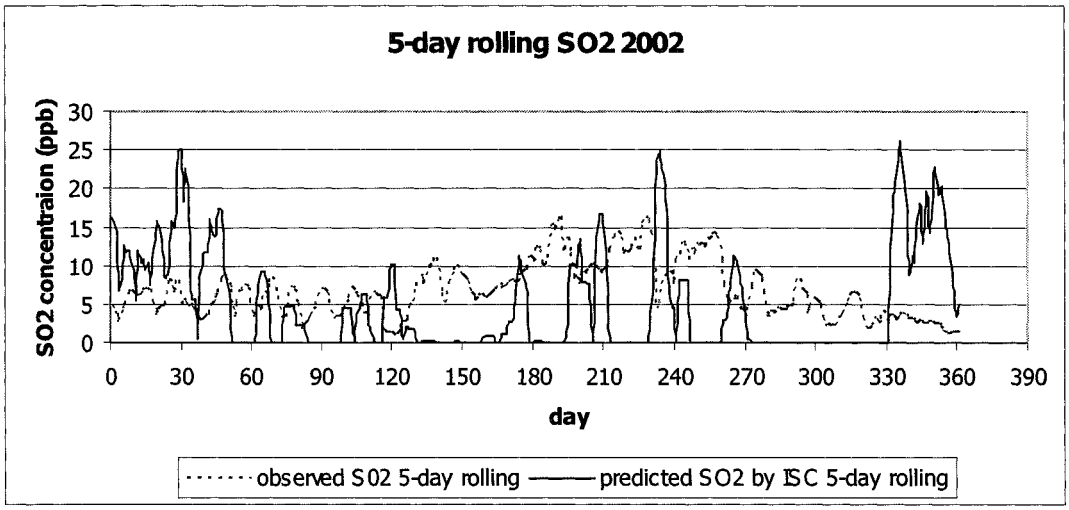


Figure 5.6a Comparison of daily average SO_2 concentrations at the 29T ambient air quality monitoring station for 2002 with those predicted by ISC.

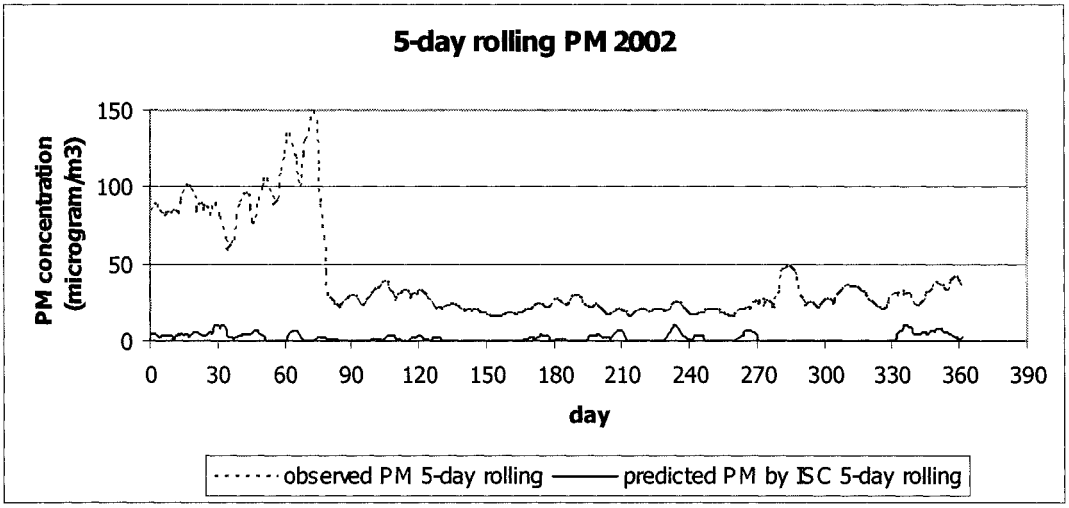


Figure 5.6b Comparison of daily average PM_{10} concentrations at the 29T ambient air quality monitoring station for 2002 with those predicted by ISC.

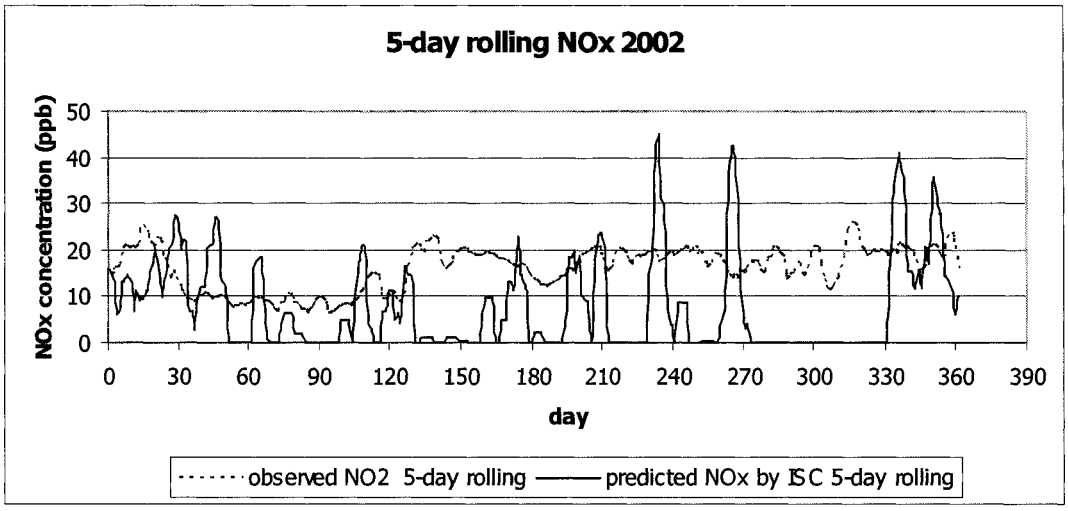


Figure 5.6c Comparison of daily average NO_x concentrations at the 29T ambient air quality monitoring station for 2002 with those predicted by ISC.

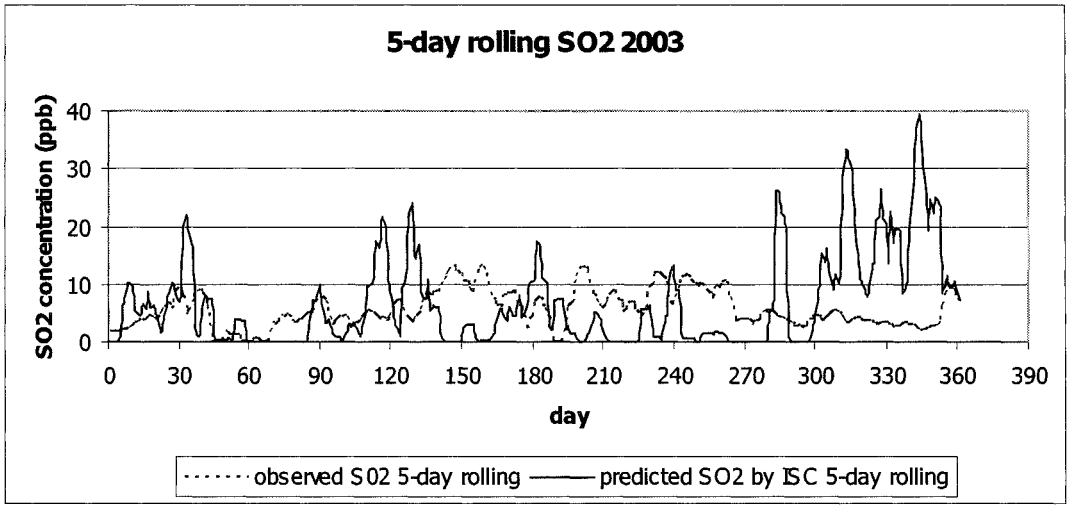


Figure 5.7a Comparison of daily average SO_2 concentrations at the 29T ambient air quality monitoring station for 2003 with those predicted by ISC.

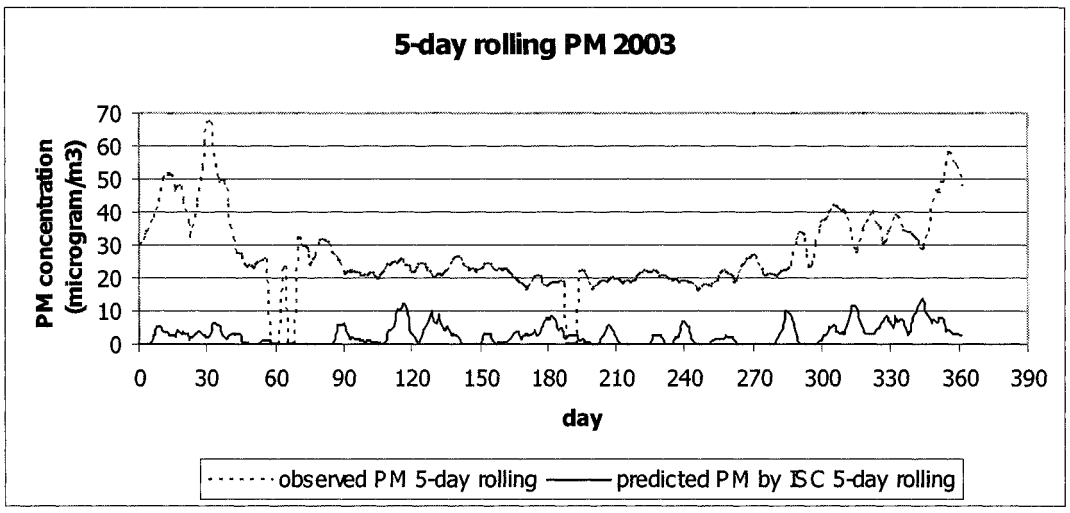


Figure 5.7b Comparison of daily average PM_{10} concentrations at the 29T ambient air quality monitoring station for 2003 with those predicted by ISC.

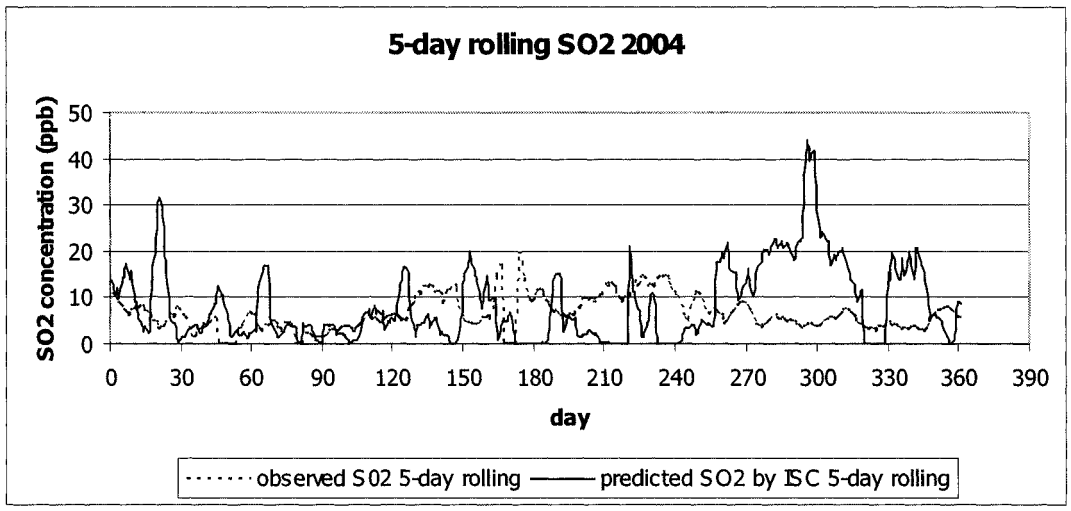


Figure 5.8a Comparison of daily average SO_2 concentrations at the 29T ambient air quality monitoring station for 2004 with those predicted by ISC.

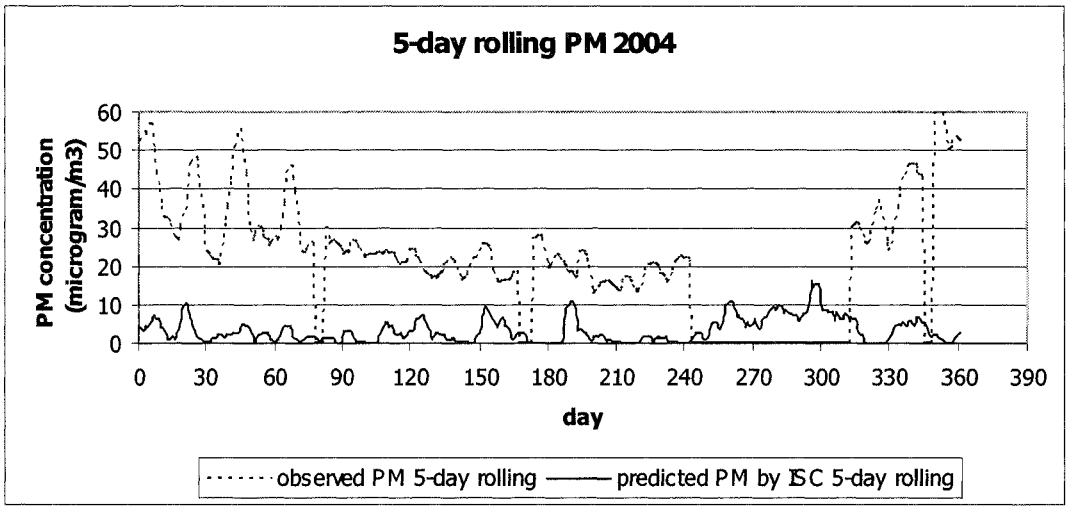


Figure 5.8b Comparison of daily average PM_{10} concentrations at the 29T ambient air quality monitoring station for 2004 with those predicted by ISC.

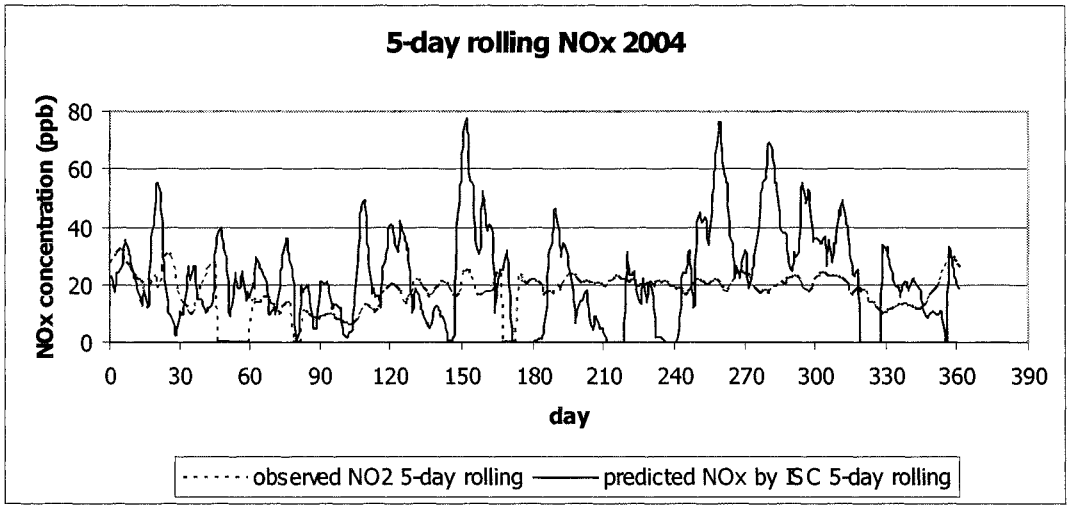
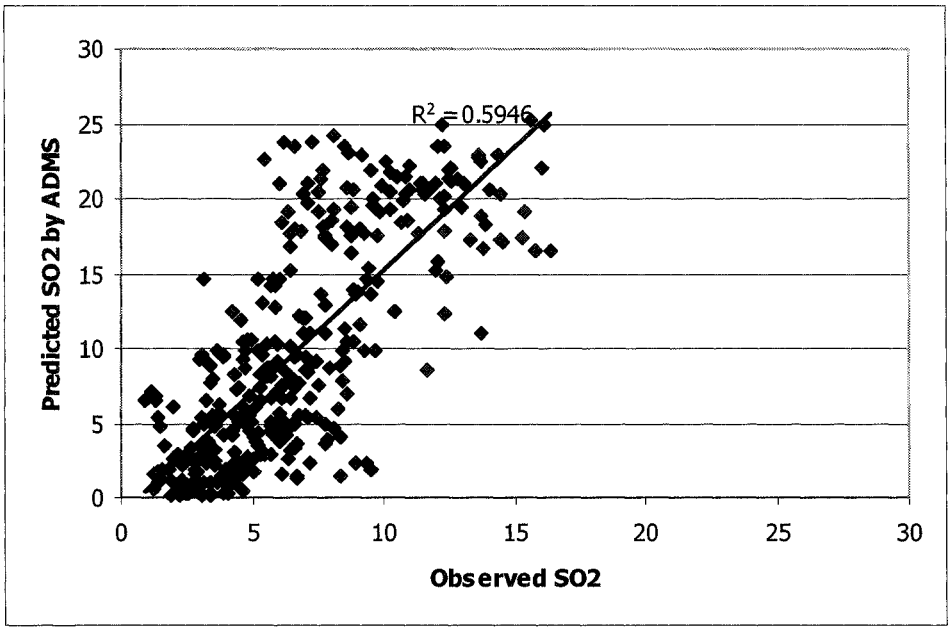
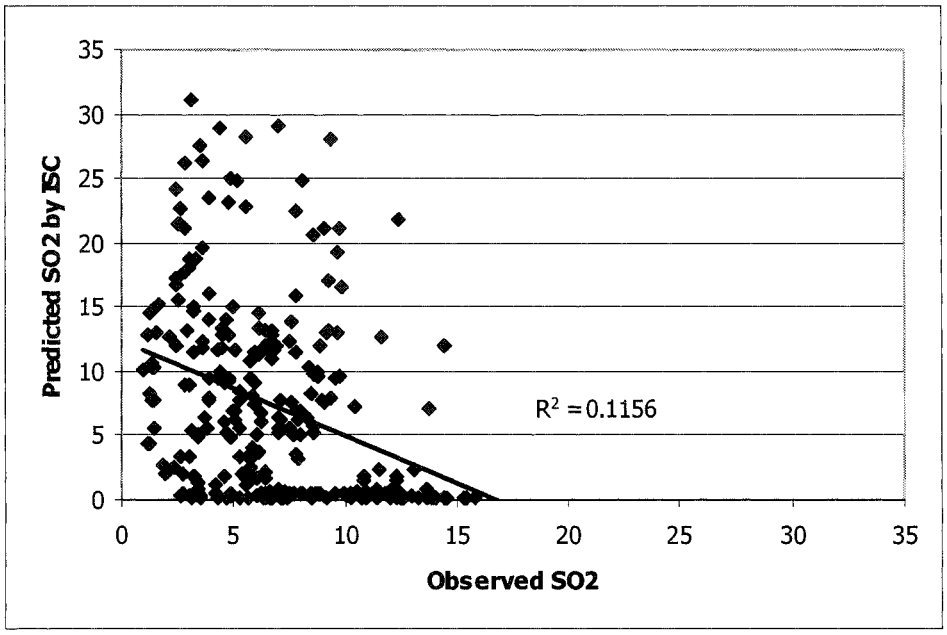


Figure 5.8c Comparison of daily average NO_x concentrations at the 29T ambient air quality monitoring station for 2004 with those predicted by ISC.

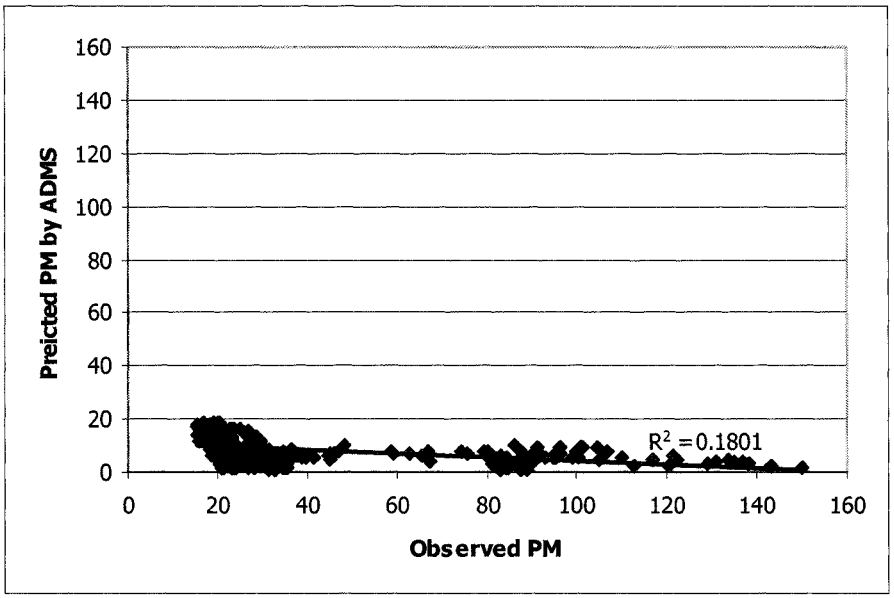


a)

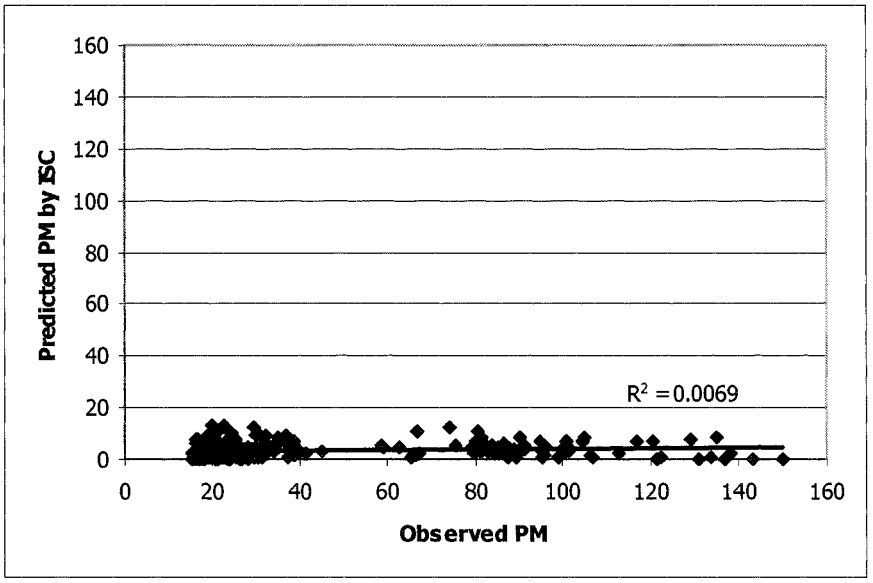


b)

Figure 5.9 Correlation between observed and modelled daily average SO_2 concentrations over a three year period for: (a) ADMS (2003 – 2005) and b) ISC (2002 – 2004).

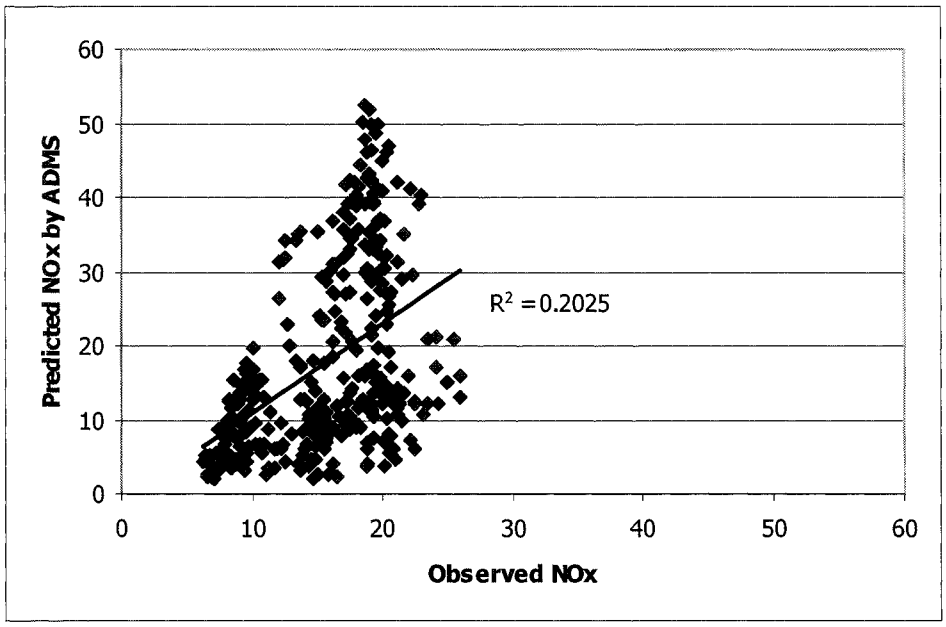


a)

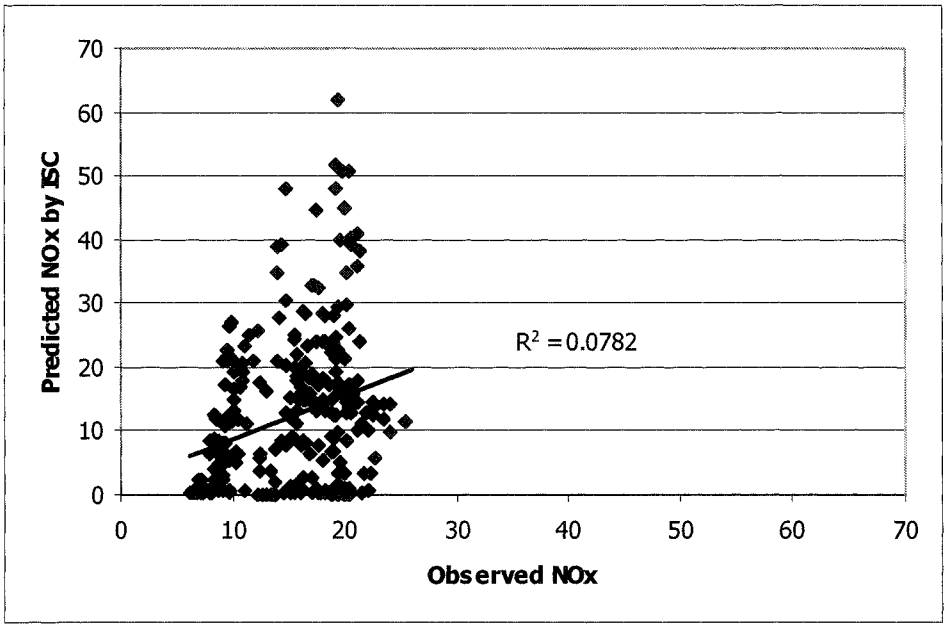


b)

Figure 5.10 Correlation between observed and modelled daily average NO_2 concentrations over a three year period for: (a) ADMS (2003 – 2005) and b) ISC (2002 – 2004).



a)



b)

Figure 5.11 Correlation between observed and modelled daily average NO_x concentrations over a three year period for: (a) ADMS (2003 – 2005) and b) ISC (2002 – 2004).

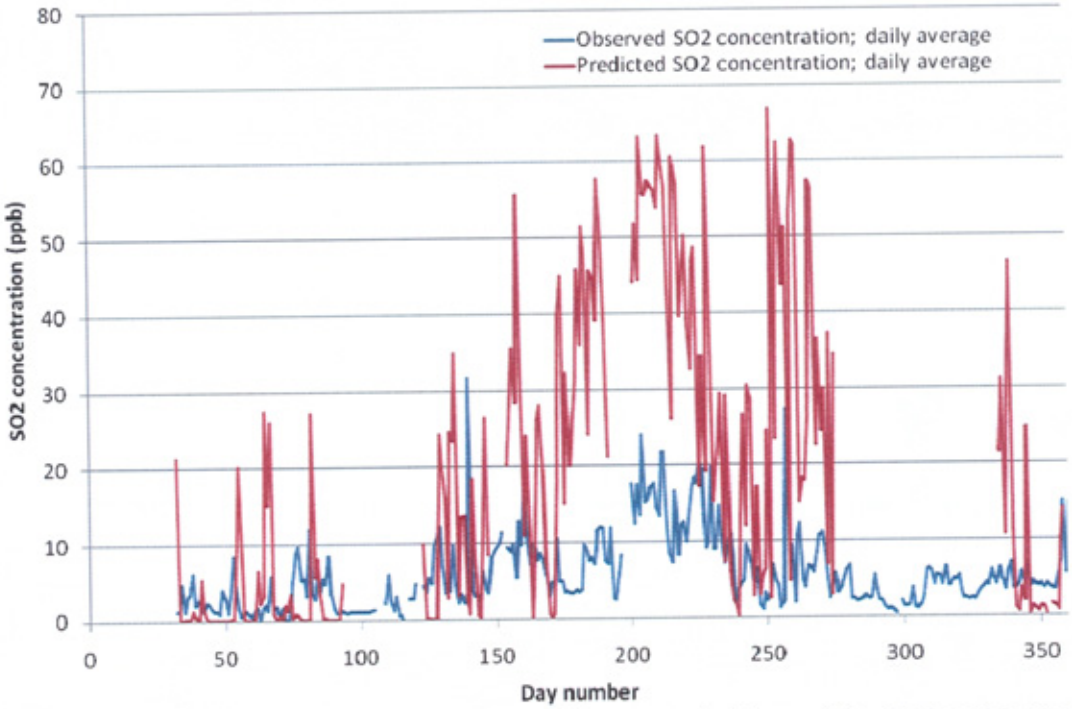


Figure 5.12a Comparison of daily average SO_2 concentrations at the 29T ambient air quality monitoring station for 2005 with those predicted by PanEIA.

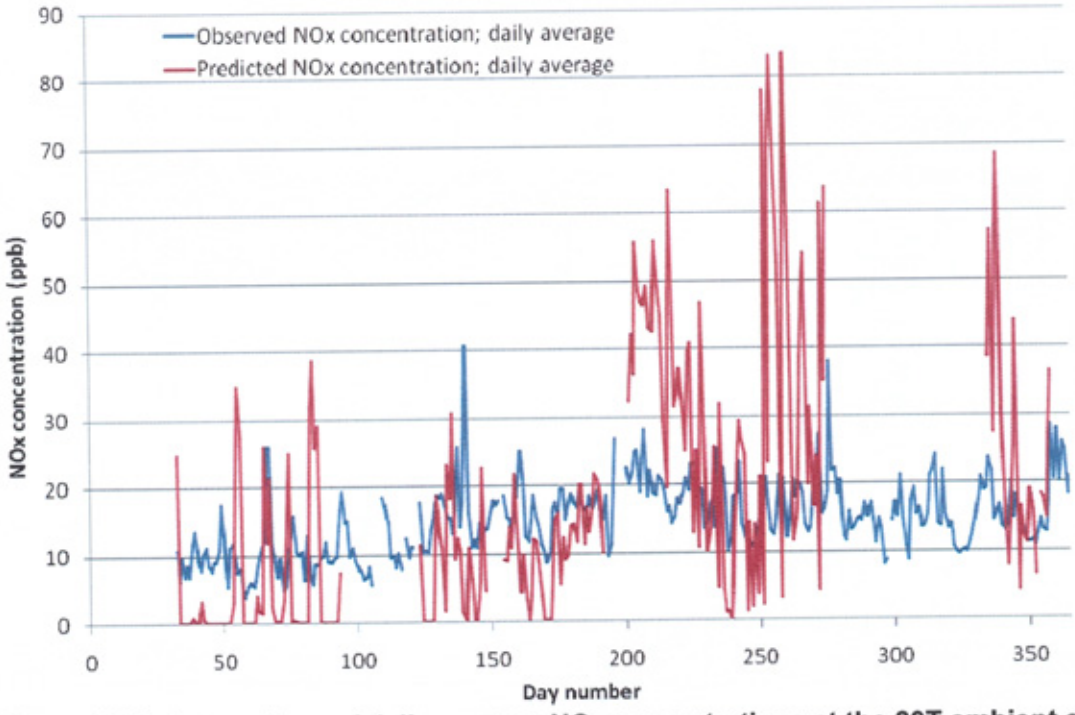


Figure 5.12b Comparison of daily average NO_x concentrations at the 29T ambient air quality monitoring station for 2005 with those predicted by PanEIA.

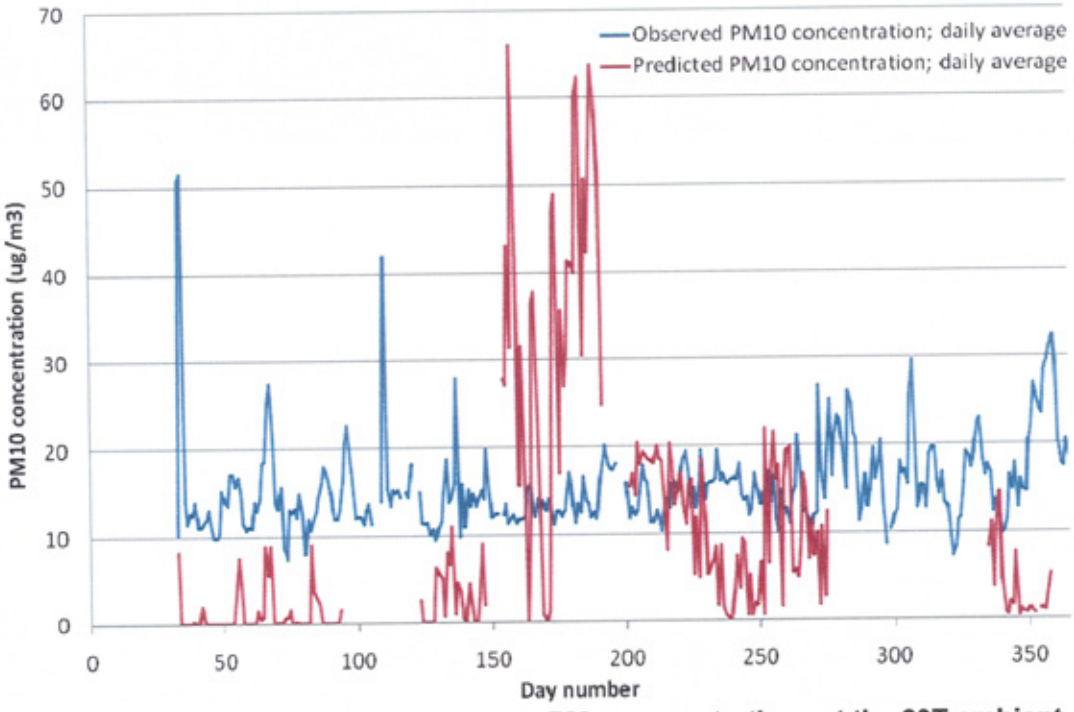


Figure 5.12c Comparison of daily average PM_{10} concentrations at the 29T ambient air quality monitoring station for 2005 with those predicted by PanEIA.

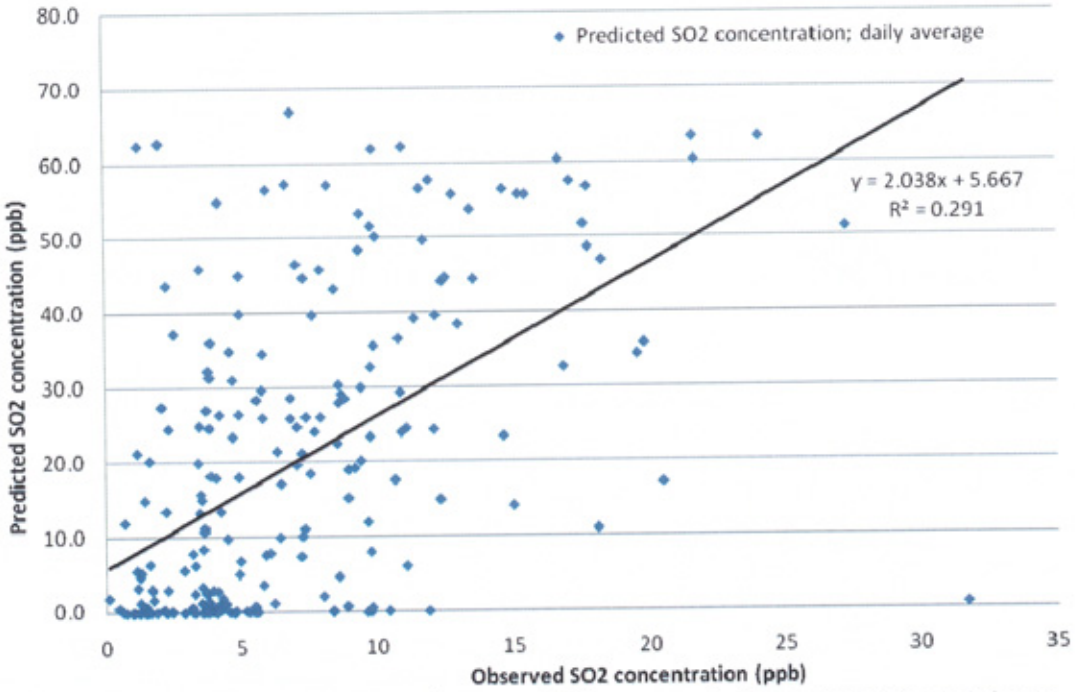
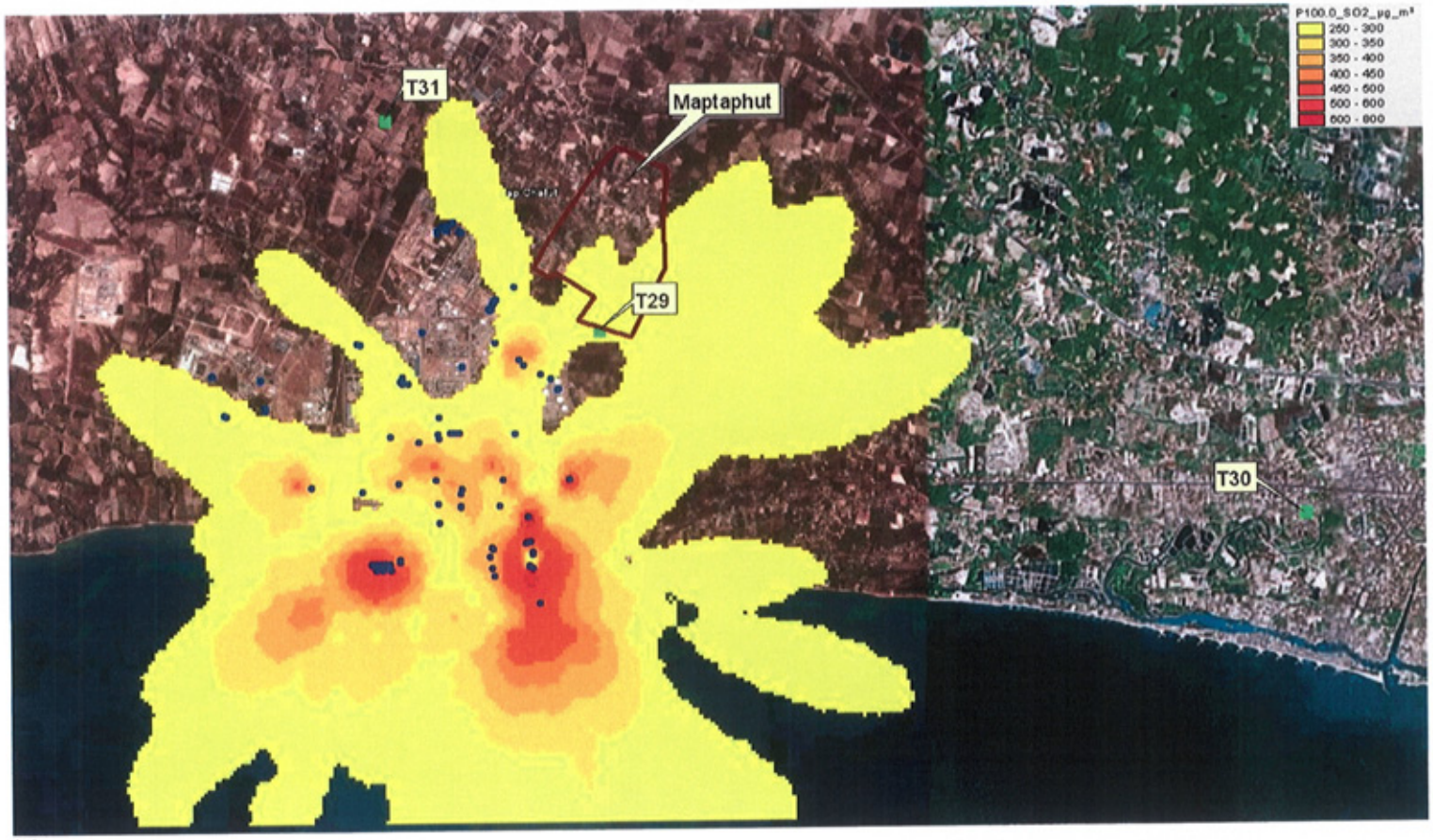
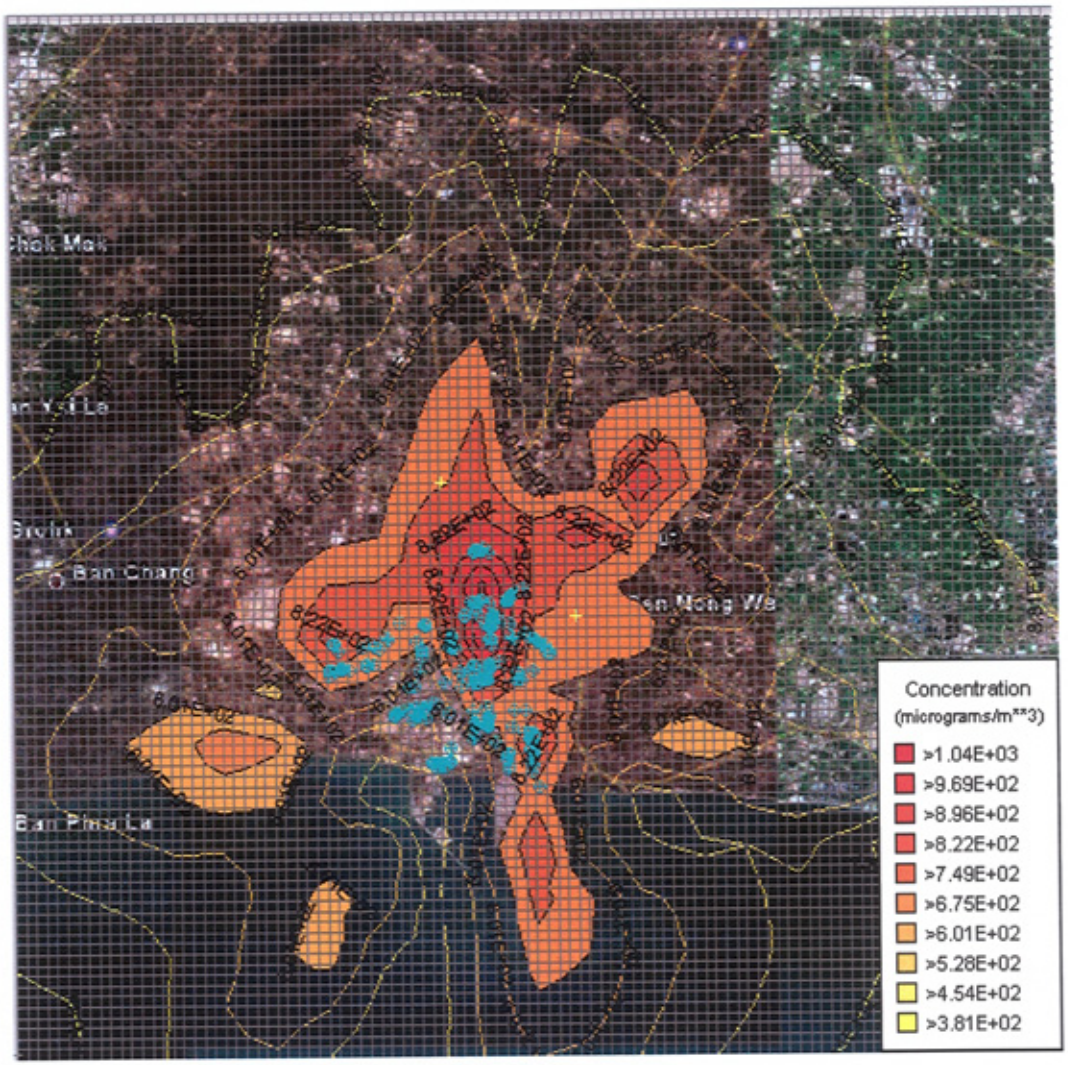


Figure 5.13 Correlation between observed SO_2 concentrations for 2005 and those predicted by PanEIA.



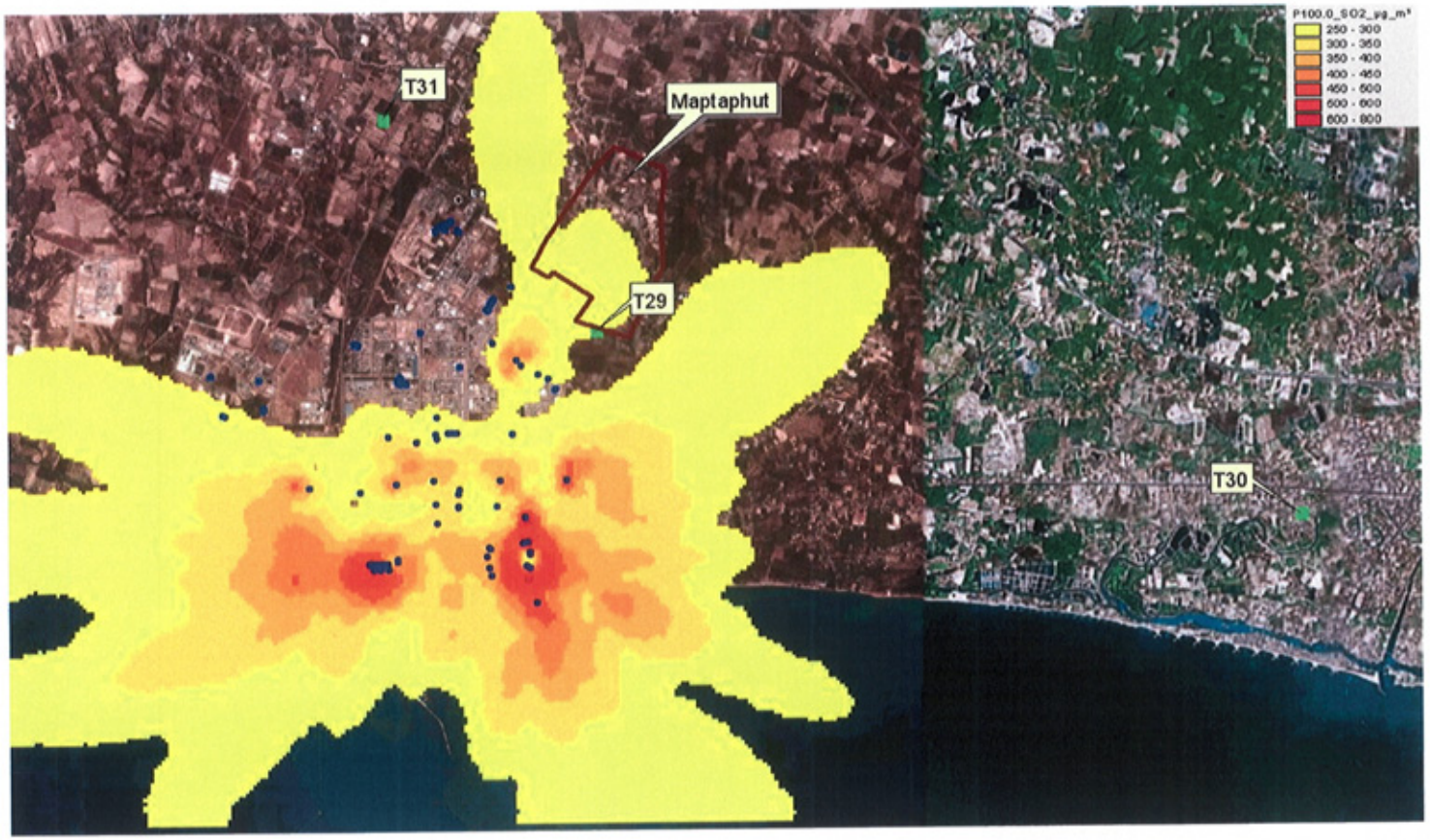
(a)
Note: blue circle points are industrial point sources, and green square points are ambient air quality stations

Figure 5.14 One hour maximum concentration contour plot of SO_2 at Maptaphut 2002 (a) ADMS result and (b) ISC result.



(b)
Note: blue circle points are industrial point sources, and yellow plus points are ambient air quality stations

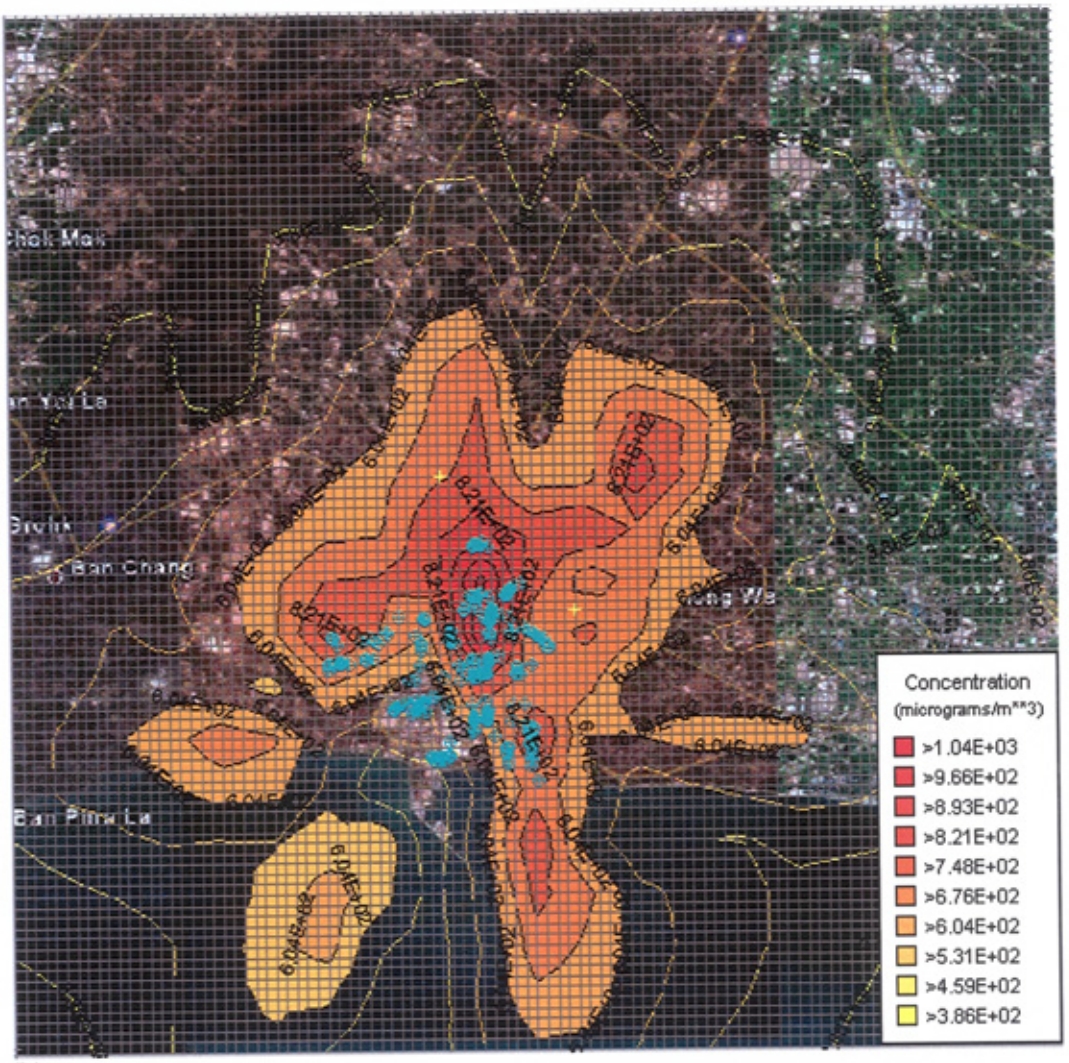
Figure 5.13 One hour maximum concentration contour plot of SO_2 at Maptaphut 2002 (a) ADMS result and (b) ISC result (continued).



(a)

Note: blue circle points are industrial point sources, and green square points are ambient air quality stations

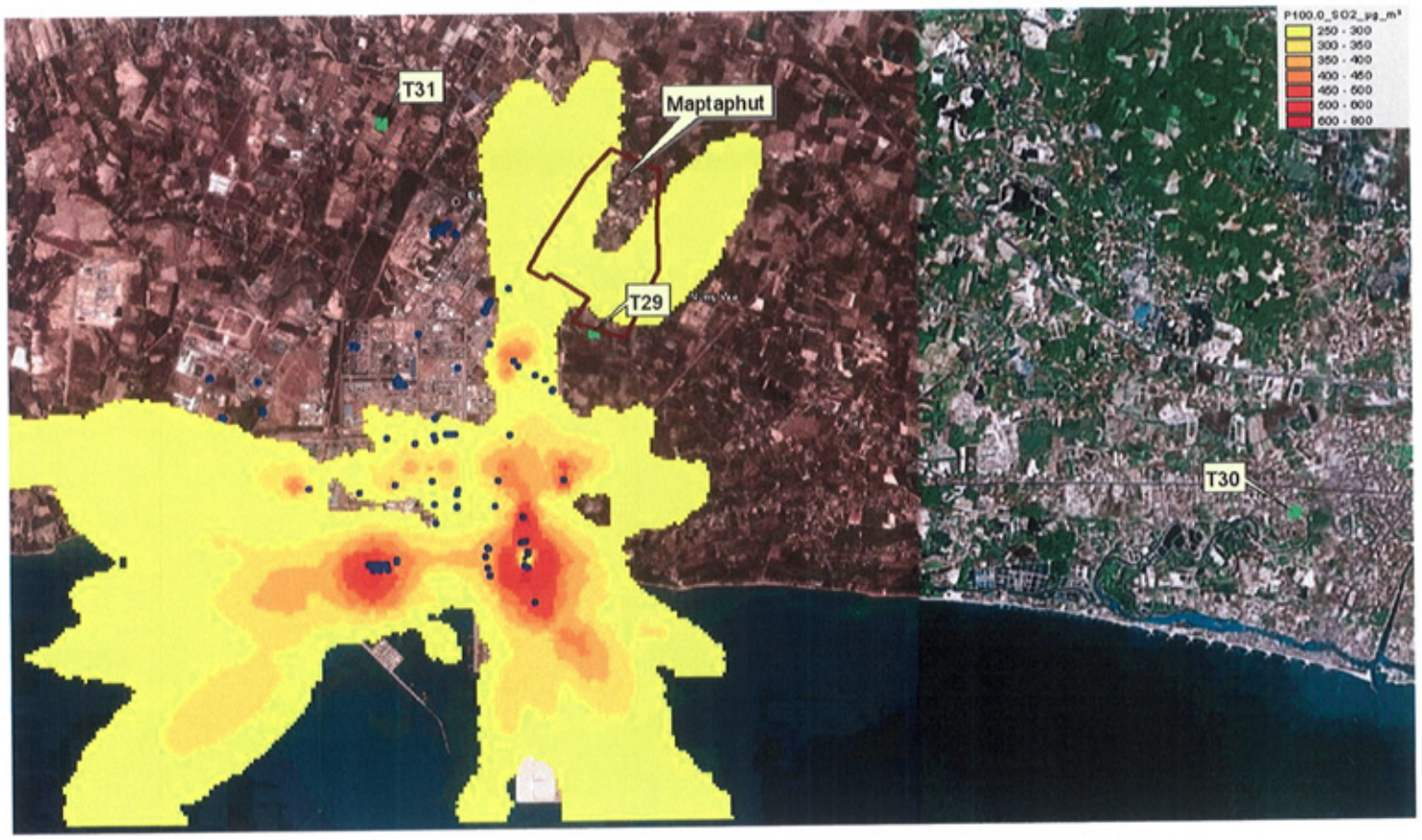
Figure 5.14 One hour maximum concentration contour plot of SO_2 at Maptaphut 2003 (a) ADMS result and (b) ISC result.



(b)

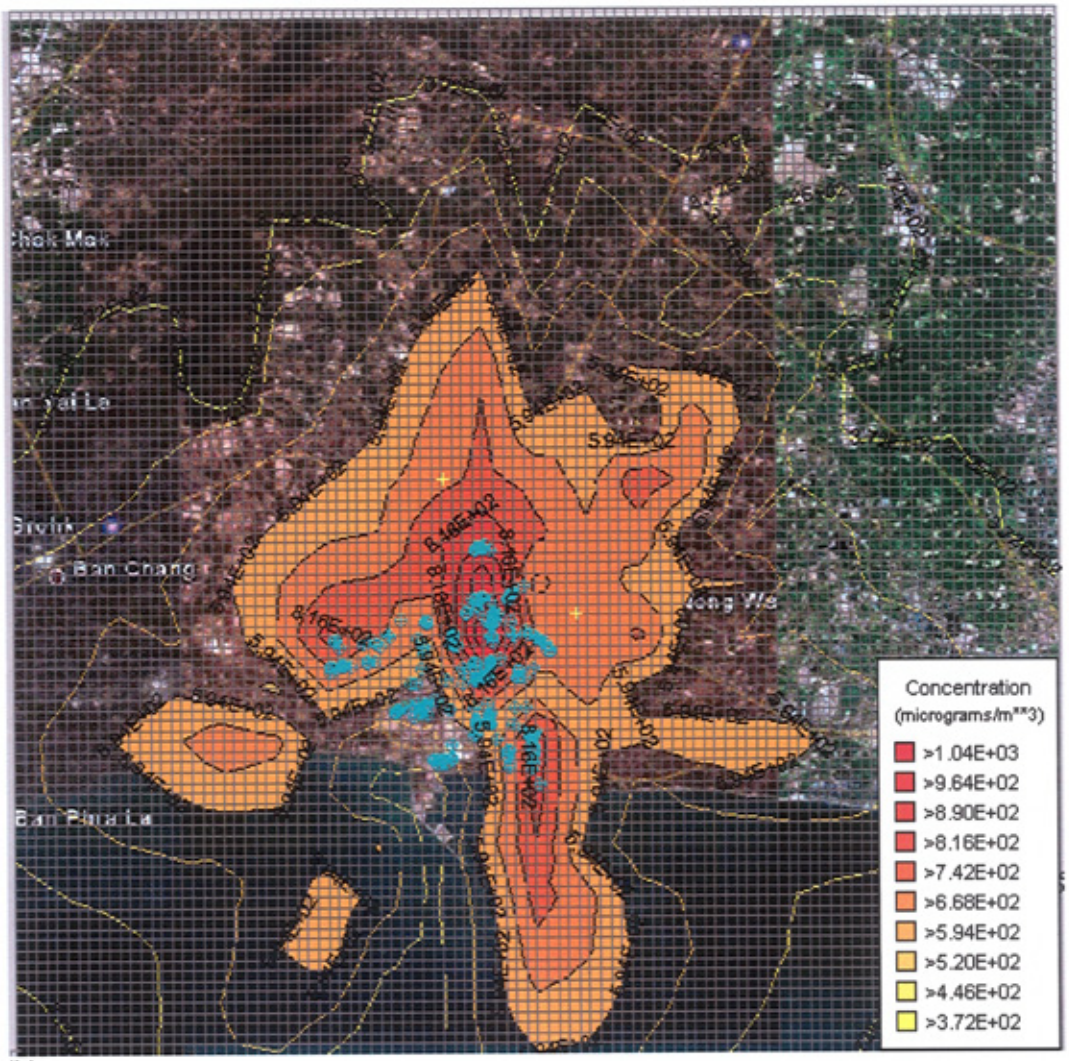
Note: blue circle points are industrial point sources, and yellow plus points are ambient air quality stations

Figure 5.14 One hour maximum concentration contour plot of SO_2 at Maptaphut 2003 (a) ADMS result and (b) ISC result (continued).



(a)
Note: blue circle points are industrial point sources, and green square points are ambient air quality stations

Figure 5.15 One hour maximum concentration contour plot of SO_2 at Maptaphut 2004 (a) ADMS result and (b) ISC result.



(b)

Note: blue circle points are industrial point sources, and yellow plus points are ambient air quality stations

Figure 5.15 One hour maximum concentration contour plot of SO_2 at Maptaphut 2004 (a) ADMS result and (b) ISC result (continued).

5.3 Results of statistical models for PM_{10} and O_3

The benefit of the dispersion models discussed in the last section is that, providing they are validated for the pollutant in question in a comparable geographical area, they can be used to: (a) predict health effects by modelling short and long term concentrations and compare these to published guidelines and standards; (b) predict improvements in ambient pollutant levels due to proposed abatement technology or traffic regulation; and (c) predict short term pollutant levels, provided that accurate meteorological forecast data is available. Moreover, the predicative capabilities should be independent of the location of the monitoring point, providing that all of the sources have been correctly characterised. The present research has shown that for SO_2, which arises only from the industrial estate, the predictive capability of ADMS is good, whilst for PM_{10} and NO_x, where significant contributions to the pollutant load arise from sources other than the industrial estate the predictions are relatively poor.

Advanced dispersion modelling software packages such as ADMS, whilst capable of producing accurate predictions, are nevertheless expensive, both in terms of the software itself and the training and maintenance that is required; hiring consultants to carry out modelling work is also an expensive option. Partly because of this, but also because of a belief that other approaches may offer more accurate predictive capability, there has been considerable research devoted to statistical methods for predicting air pollution levels. Recent examples have included: a multiple linear regression (MLR) model for predicting total suspended solids and SO_2 levels in Turkey (Ilten & Selici, 2008); MLR applied to PM_{10} forecasting in three Austrian cities (Stadlober *et al.*, 2008) ; MLR and neural networks used to predict next-day hourly ozone concentrations in Oporto, Portugal (Sousa *et al.*, 2007) (Sousa *et al.*, 2006); MLR and ARIMA (autoregressive and integrated moving average) applied to prediction of NO_2 and SO_2 in Salamanca, Spain (Panero *et al.*, 1997); ozone prediction in Castilla-Leon, Spain, using principal component analysis (Alvarez *et al.*, 2000); MLR applied to carbon monoxide levels in Northern Italy (Peace *et al*, 2005); stepwise MLR applied to the prediction of ozone maxima several hours in advance in Errenteria, Spain (Barrero *et al.,* 2006); MLR, ARIMA and MLR/ARIMA combined to predict particulate matter concentrations in Delhi and Hong Kong, based on meteorological parameters only (Goyal, *et al.*, 2006).

The benefits of statistical models are that they are relatively cheap to use, once developed, and that they can be associated with a high degree of accuracy. In addition, the models do not require the sources to be characterised, merely that pollutant emissions are constant, or that in cases of seasonal or daily variability that this is taken into account; for example by splitting the dataset into seasons, weekdays, weekends etc and deriving separate equations for each of the sub- sets (Stadlober *et al.,* 2008). The downside is that such models are usually location specific and not readily transferable to other settings (for which a new equation would have to be developed). Thus, such approaches might be used to predict concentrations next to a specific road source, or near to a specific industrial estate, as is the case with this work. Moreover, if there was no ambient monitoring station near to the point of interest, then a mobile station would be required to collect data against which the model could be derived and validated.

This section presents the results of some stepwise multiple linear regression studies aimed at producing an accurate predictive model for the two pollutants associated with the most significant health effects, ozone and PM_{10} (WHO, 2005). These are also pollutants that were either not attempted for modelling using the dispersion models (ozone) or produced poor results (PM_{10}).

A useful statistical model should be able to predict / forecast pollutant concentrations based on either measured, or accurately forecast meteorological conditions; in this way only general meteorological conditions for an area are required to make the predictions. However, it is often found that in addition to meteorological data, pollutant concentrations, either measured or predicted, also need to be input into the model in order to improve performance; for example Sousa's study in Oporto where hourly ozone concentrations were predicted one day in advance on the basis of the previous day's temperature (T), relative humidity (RH), wind velocity (V), $[NO_2]$ and $[O_3]$ (Sousa *et al.*, 2007).

The present study has derived equations for both O_3 and PM_{10} that are based on meteorological data *and* pollutant levels. Currently these equations give instantaneous predictions, i.e. given the current hour's meteorological and pollution data, the PM_{10} and ozone concentration can be predicted. In some respects this has limited usefulness, since the stations that monitor the pollutants concentrations that are required for input into the model are also likely to monitor ozone and PM_{10} (though if successful, the use of a more restricted, cheaper, monitoring suite could be proposed at some sites, relying on the

statistical techniques to derive the other values). Nevertheless, this type of study does have value in determining the principal factors contributing towards ozone formation in this geographical location; similar literature studies have been carried out, for example a recent study using stepwise MLR that determined the principal meteorological and pollutant contributions towards ozone formation in the daytime and night-time in Kuwait (Abdul-Wahab *et al.*, 2005).

The statistical model was developed using the Multiple Linear Regression method (MLR). The equations are developed using observed data for a 5-year period for 29T, 30T and 31T ambient monitoring stations during the dry season. The model can predict *maximum* 1-h PM_{10} and ozone concentrations based on meteorological parameters and other pollutants. In a sense, because the model has been derived using data from all three of the regional ambient air quality monitoring stations it can be considered to be applicable to all locations, thus removing one of the criticisms raised previously, i.e. that statistical models tend to be location specific.

Table 5.2 shows the determined parameters for the MLR using SPSS. Both unstandardised and standardised parameters are shown, the latter of which compares the *relative* contribution of the various parameters; additionally, the t values tests whether the regression coefficients significantly differ from zero. Table 5.2 clearly shows that WS50 is the most significant factor contributing towards determining the PM_{10} concentration, followed by WS100, T2M, WS10, RH and CO. For ozone, RH is the most significant factor, followed by NMHC, WH10, CH4, NO and NO_2. From these results a statistical model was generated to predict the highest hourly PM10 and O3 concentration.

5.3.1 Statistical Model for Daily Maximum of 1-h PM_{10}

For PM_{10}, the regression was performed for the maximum daytime 1-h PM_{10} during the day, (6:00 am to 10:00am) and the pollutant concentrations of CO, O_3, SO_2, NO, NO_2, CH_4, and NMHC, and the above listed meteorological parameters recorded at the same time as when maximum PM_{10} occurred. The model obtained is as follows:

$$PM_{10} = 305.8 + 26.6\ CO - 0.7\ RH - 5.3\ T + 27.1\ WS10 - 52.1\ WS50 + 24.9\ WS100 \qquad \text{Equation 5.1}$$

The model statistics presented in Table 5.2 shows the regression model for PM_{10} is strongly significant at 0.01 significance level. However, the model has a low coefficient of

determination (R^2), 21%, and rather high standard errors. Nevertheless, the model demonstrates that PM_{10} is positively correlated with CO, indicating both pollutants arise from the traffic sources. Stadlober et al, who have developed a daily forecasting model for PM_{10} in Austrian cities found that wind speed, precipitation, the presence of an inversion layer and a lag factor for PM_{10} concentration from the previous day were important determinants in the accuracy of the forecasts; they found that 73% of PM_{10} forecasts were 'satisfactory' or better (Stadlober *et al.,* 2008). Goyal et al, who used both ARIMA and MLR to predict daily 24-hour averaged particulate concentrations in Delhi and Hong Kong, were able to predict concentrations entirely from meteorological data, with wind speed, solar radiation and surface temperature found to be the important contributing factors; the model gave an average error between observed and predicted concentrations of 25% (Goyal *et al.,* 2006).

5.3.2 Statistical Model for Daily Maximum of 1-h O_3

Early morning measurements are often used for evaluating emission inventories because chemical losses are relatively small at that time and ambient concentrations reflect the ratio of directly emitted species (Sillman, 1999). The daily maximum hourly O_3 value, which is normally observed at noon and early afternoon, was selected from the period of 12:00-16:00. Stepwise regression was performed on the daily maximum 1-h O_3 and the pollutant concentrations of CO, PM_{10}, SO_2, NO, NO_2, CH_4, and NMHC at 7:00am and the meteorological parameters at 12:00. The model obtained is as follows:

$$O_3 = 74.1 - 0.7\ NO + 0.5\ NO_2 + 3.4\ CH_4 + 10.7\ NMHC - 0.6\ RH - 5.6\ WS10 \qquad \text{Equation 5.2}$$

The coefficient of determination (R^2) for the O_3 model is 48%, which is higher than the PM_{10} model. The regression model is strongly significant at 0.01 significance level. O_3 is shown to positively correlate with its precursors of NMHC and NO_2 and negatively correlate with NO, which should be expected from the O_3 chemistry. In a comparable study, Abdul-Wahab *et al* used MLR and Principal Component Analysis to model day and night-time ozone concentrations in Kuwait City. The day-time equation, which gave an R^2 value of 0.82, had NO concentration, temperature, solar radiation and SO_2 concentration and the most significant determinants, with NO and SO_2 both being negatively correlated (Abdul-Wahab *et al.,* 2005). Barerro *et al*, who used stepwise MLR to predict ozone maxima 7 hours in advance using a combination of meteorological data and pollution concentrations (solar insolation, day number and 7-hour delayed NO, NO_2 and O_3 concentrations); the model performed well, with an R^2 of 0.6 and with 50% of all predicted

values lying within 15% of the observed (Barrero *et al.,* 2006). I n contrast to these results Sousa *et al*, as discussed above, found that NO was *not* a significant determinant, but that the previous day's NO_2 and ozone concentration, together with the previous day's relative humidity and wind speed were significant in forecasting the following day's ozone concentration (Sousa *et al.*, 2007).

Finally a comment on wind direction, which wasn't a significant determinant in the equations derived for this study, nor for the majority of the literature studies. Nevertheless it *should* be important for pollutant dispersion, as indicated in the directional analysis results in section 5.2. Treatment of wind direction in linear regression models presents some difficulties because this parameter, like Julian day number, is a circular, not linear variable; as an illustration, the average of 350° and 10° is 360°, not the 180° that would be calculated if they were treated as linear variables. There are transformations that can be carried out so that circular variables can be included in MLR studies; Jammalamadaka and Lund (2006) have outlined just such an approach for modelling urban ozone levels in Texas. Peace *et al* (2005) have also suggested used a transformation in order to treat wind direction as part of their study on roadside carbon monoxide levels in Northern Italy. An extension of this work may be to test the significance of wind direction.

5.3.2 Comparison of observed and predicted ozone and PM10 concentrations

Comparison between the observed and model-predicted concentrations for ozone and PM_{10} at 31T station (dry season, 1998 to 2002) are shown in Figures 5.16 and 5.17 respectively.

There is a better agreement between the observed and predicted concentrations for O_3 than for PM_{10}. The models underestimated the peak values of both pollutants, especially for PM_{10}. Statistical models are normally not capable of predicting peak values and the models obtained in this study are not exceptional. An improvement of the statistical prediction can be made if the meteorology was first classified into clusters with homogeneous conditions.

Table 5.2 Summary of multiple linear regression equations for PM_{10} and O_3

Model	R	**R^2 (%)**	Standard Error of Estimate		
PM_{10}	0.459	21	52.9 (μg m^{-3})		
O_3	0.689	48	15.2 (ppb)		
Model	**Variable**	**Unstandardised Coefficient(B)**	**Standardized Coefficient(Beta)**	**t statistics**	**Significance**
PM_{10}	(Constant)	305.8		6.0	0.000
	WS50	-52.1	-1.6	-7.2	0.000
	WS100	24.9	1.0	5.1	0.000
	RH	-0.7	-0.2	-2.8	0.005
	T2M	-5.3	-0.3	-3.9	0.000
	WS10	27.1	0.3	3.7	0.000
	CO	26.6	0.1	2.0	0.041
O_3	(Constant)	74.1		13.4	0.000
	RH	-0.6	-0.5	-10.1	0.000
	CH_4	3.4	0.2	3.6	0.000
	NMHC	10.7	0.2	4.4	0.000
	WS10	-5.6	-0.2	-3.7	0.000
	NO	-0.7	-0.2	-2.9	0.005
	NO_2	0.5	0.1	2.1	0.040
PM-10	R = 0.459	, R^2 = 21% ,	, F = 12.81 ,	SE = 52.9	(μg m^{-3})
O3 R = 0.459	R = 0.689	, R^2 = 48% ,	, F = 33.7 ,	SE = 15.2 ((ppb)	(ppb)

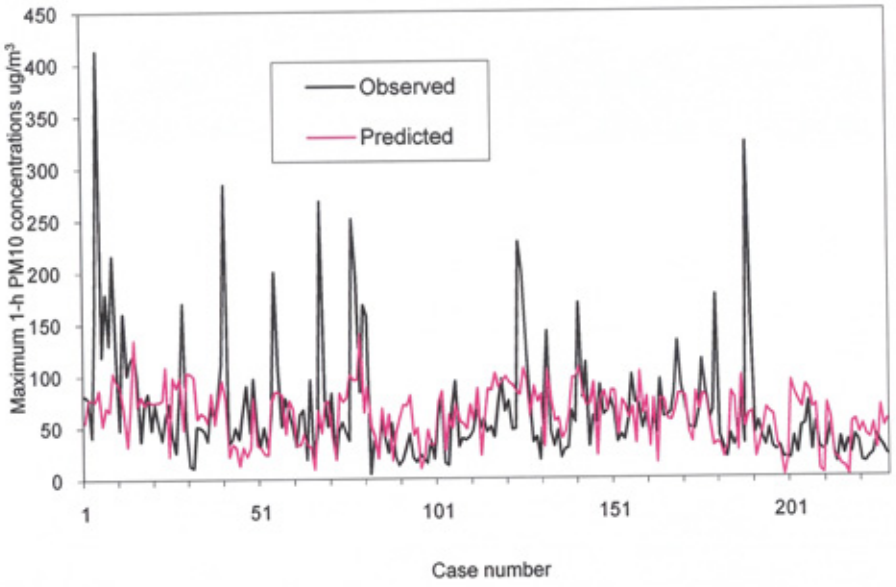


Figure 5.16 Comparison of the observed and predicted concentrations for PM_{10}.
Note: five-year average at 29, 30, and 31T

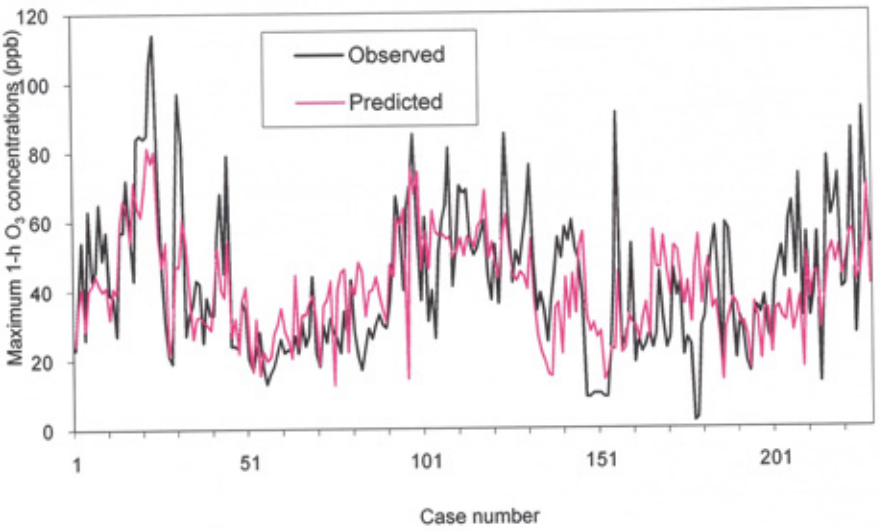


Figure 5.17 Comparison of the observed and predicted concentrations for O_3.
Note: five-year average at 29, 30, and 31T

5.3.3 Time Series Analysis of the Statistical Model

Having generated the MLR statistical models, this section compares time series' of observed against predicted concentrations at specific locations. As mentioned earlier, the equations are effectively a generic model for the Rayong region, having been derived from meteorological and pollutant monitoring data for all three of Rayong's ambient air quality monitoring stations. Here, the applicability of the model in describing maximum ozone and PM10 concentrations when using monitoring data at specific locations in Rayong province (29T and 31T near to the Maptaphut industrial estate, and 30T in Rayong City) is examined for (a) the time period from which the model was generated (1998 to 2002) (b) the next two years (2003 -2004) and (c) a region outside Rayong.

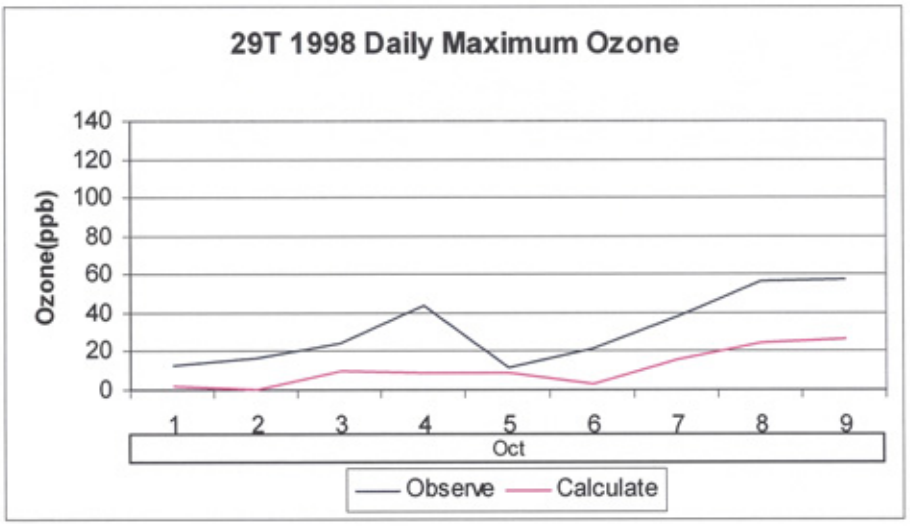
The multiple linear regression equation derived from the 5-year average during dry season (1998 – 2002) at three ambient air quality monitoring stations (29, 30, 31T) at Rayong province was used to model daily maximum Ozone and PM10 data yearly 1998, 1999, 2000, 2001, and 2002 at the same stations 29T, 30T, and 31T. The results shown in Figures 5.18 and 5.19 for ozone and PM_{10}, respectively, indicate that the general trend of concentrations is predicted, though not the peaks, as discussed in the preceding section.

Time series for the same multiple linear regression equation calculated daily maximum concentration of Ozone and PM_{10} at the same three stations (29, 30, 31T) in the subsequent two years (2003 and 2004) are shown in Figures 5.20 and 5.21 for ozone and PM_{10}, respectively; again showing similarities in trends.

Finally as a test of the applicability of the model in other regions, time series comparisons have been made with monitoring data from two ambient monitoring stations in Chonburi province, which is about 80 km from Maptaphut. The same MLR equation was applied to pollutant and meteorological data obtained from these stations and used to predict daily maximum concentrations for ozone and PM_{10} for the 2004 period. The meteorological and ambient air quality data are from two stations namely 32T and 33T year 2004, except wind speed data which had to be taken from the 100-m mast at Rayong. The calculated data of daily maximum Ozone shows good general agreement in terms of the trends (Figure 5.22) though the PM_{10} predicted concentrations are not as well described (Figure 5.23), possibly because the wind speed data has been taken from Rayong.

Looking at the average concentrations for predicted and monitored over the period for which the model has been derived, for ozone, the average observed concentration is 34.80 ppb compared to an average predicted concentration of 38.00 ppb. The average error percentage is 21 %, which means that for a predicted concentration of 100 ppb, the observed concentration could lie between 89 – 121 ppb. Further information about daily maximum monitoring data and predicted concentrations for the years 1998-2004 during dry season (not include wet season, May to September) at 29, 30, 31T station with differences between both data and error percentage is presented in Appendix D.

For PM_{10}, the average value of observed data is 64.21μg m^{-3} and of predicted concentrations is 61.78 μg m^{-3}, again quite similar. In this case the average error percentage is 38 % that means if calculation is 100 μg m^{-3}, the real data could be during 62 – 138 μg m^{-3}. The equation results are not fit with the monitoring data correctly due to a more limited set of PM_{10} monitoring data. For more accuracy, future research should determine the prediction equation using a larger number of monitoring points. More information about daily maximum monitoring data and predicted concentrations of PM10 for the years 1998-2004 during dry season (not include wet season, May to September) at 29, 30, 31T station with actual and percentage differences is given in Appendix D.



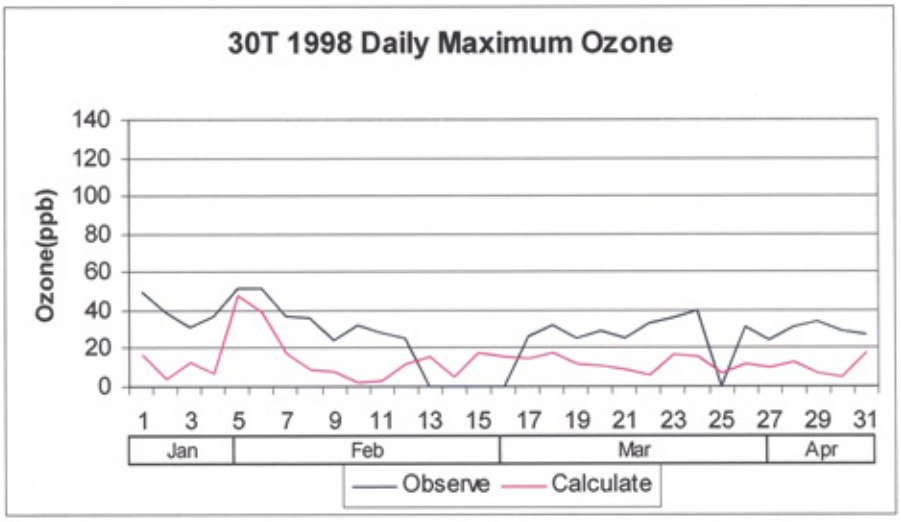
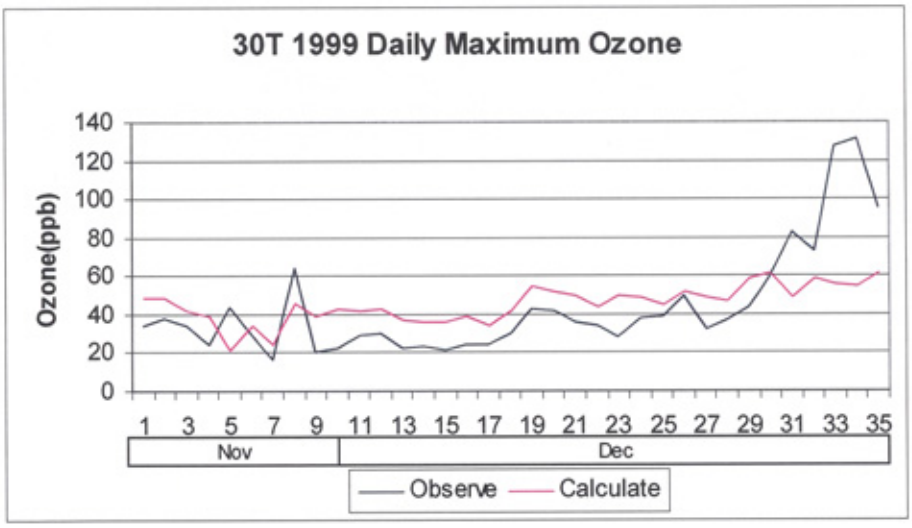


Figure 5.18 Yearly modelled daily maximum Ozone at the 29, 30, 31T station year 1998-2002.



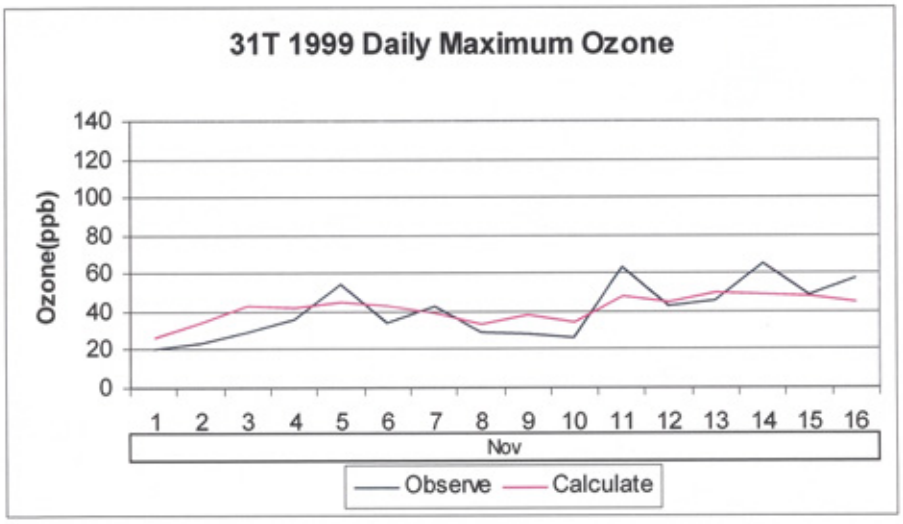
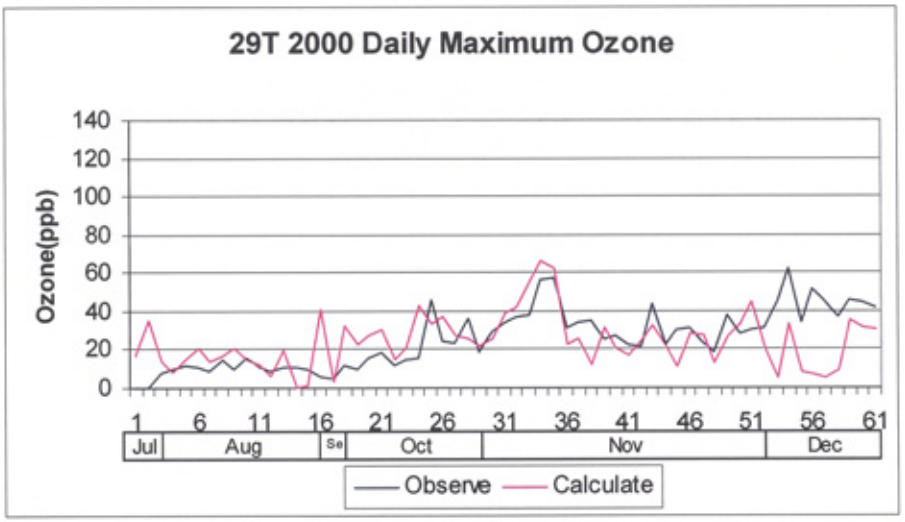


Figure 5.18 Yearly modelled daily maximum Ozone at the 29, 30, 31T station year 1998-2002 (continued).



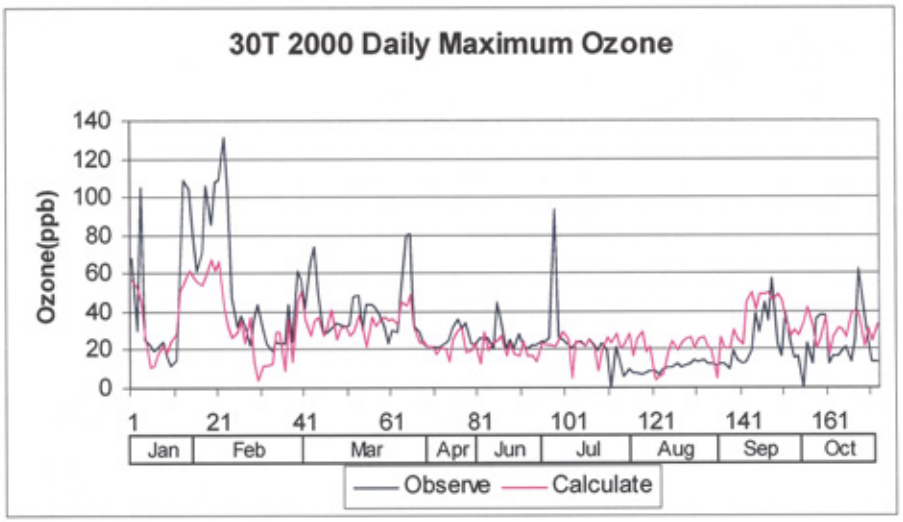
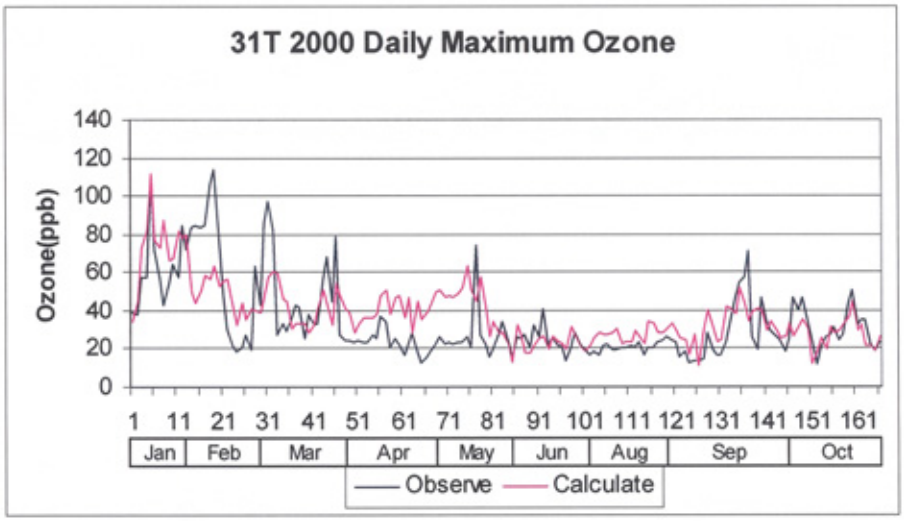


Figure 5.18 Yearly modelled daily maximum Ozone at the 29, 30, 31T station year 1998-2002 (continued).



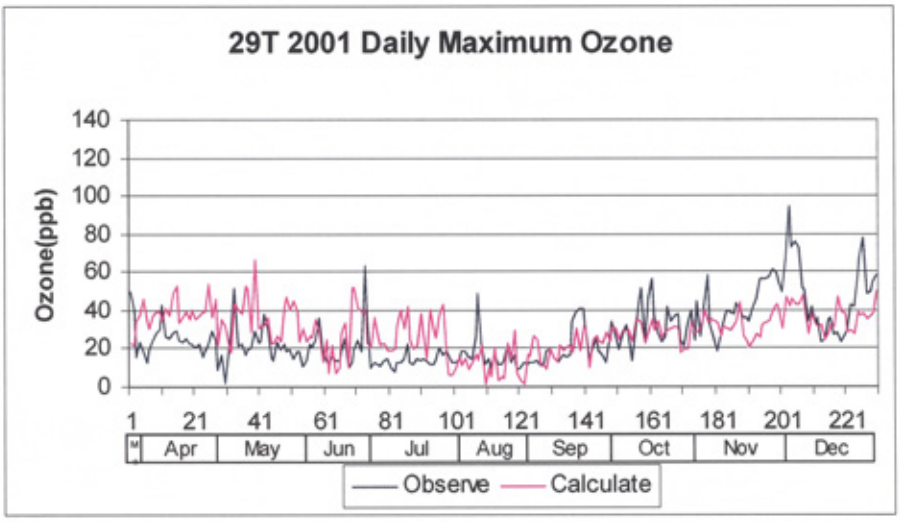
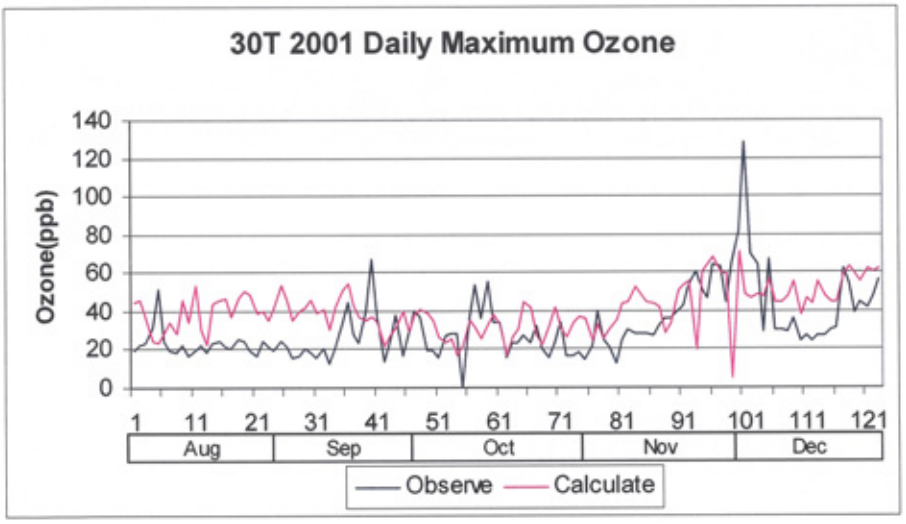


Figure 5.18 Yearly modelled daily maximum Ozone at the 29, 30, 31T station year 1998-2002 (continued).



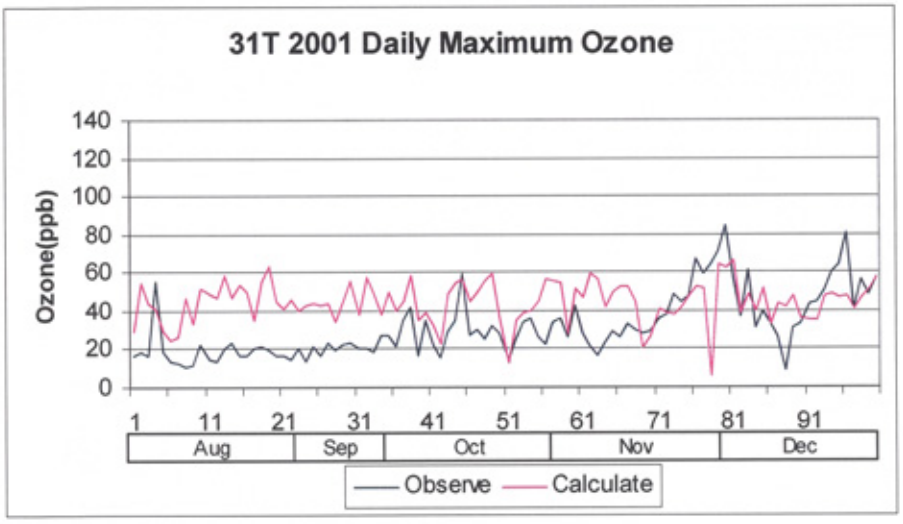
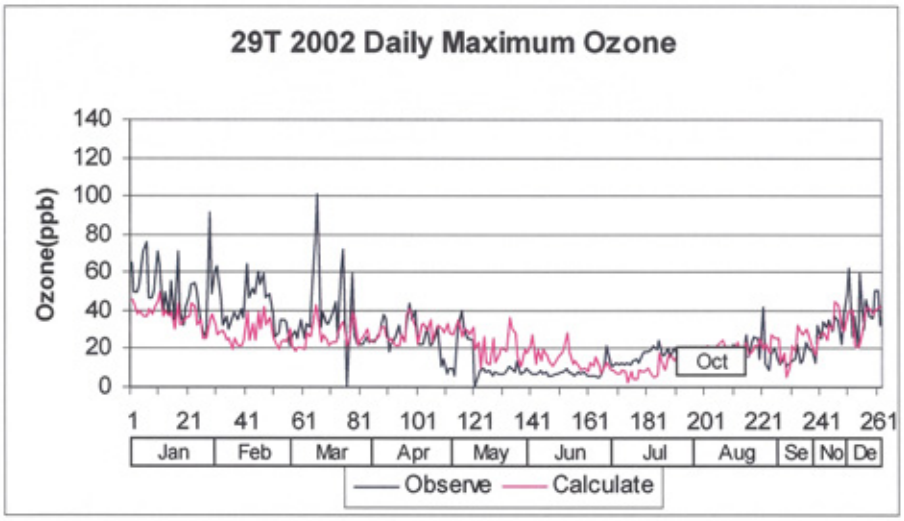


Figure 5.18 Yearly modelled daily maximum Ozone at the 29, 30, 31T station year 1998-2002 (continued).



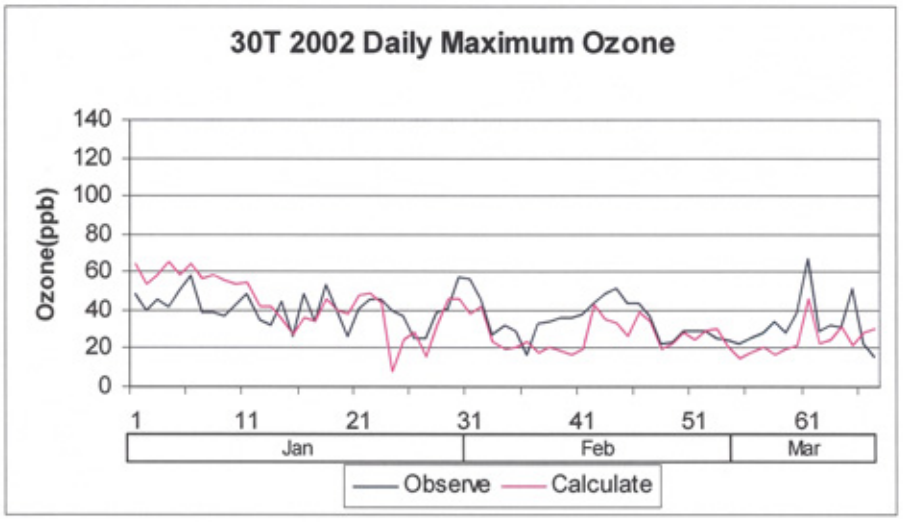


Figure 5.18 Yearly modelled daily maximum Ozone at the 29, 30, 31T station year 1998-2002 (continued).

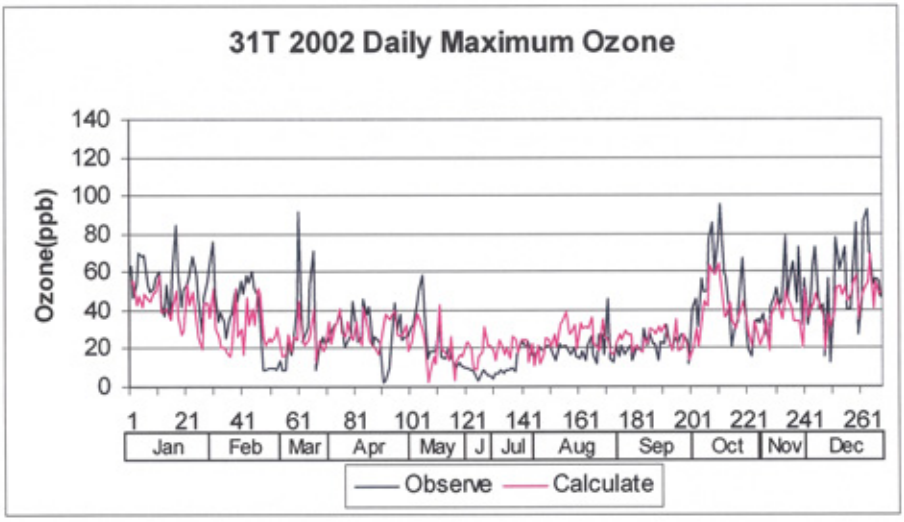
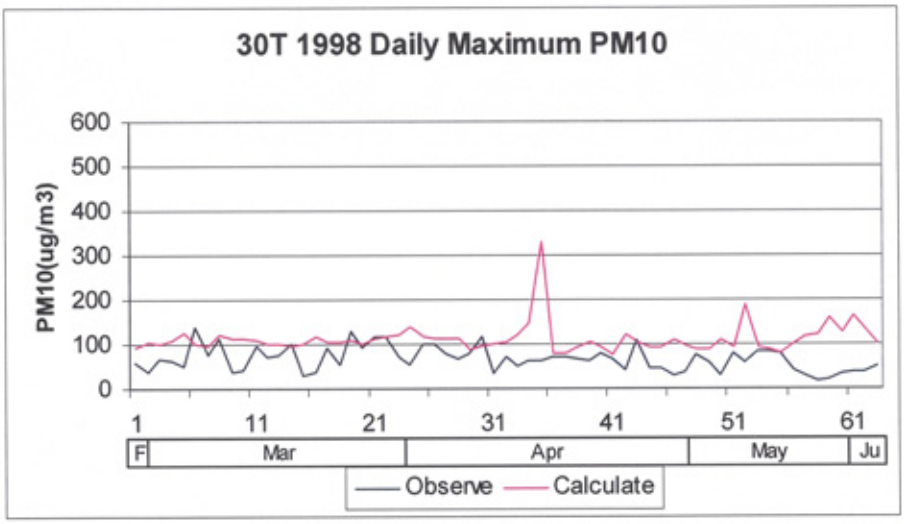


Figure 5.18 Yearly modelled daily maximum Ozone at the 29, 30, 31T station year 1998-2002 (continued).



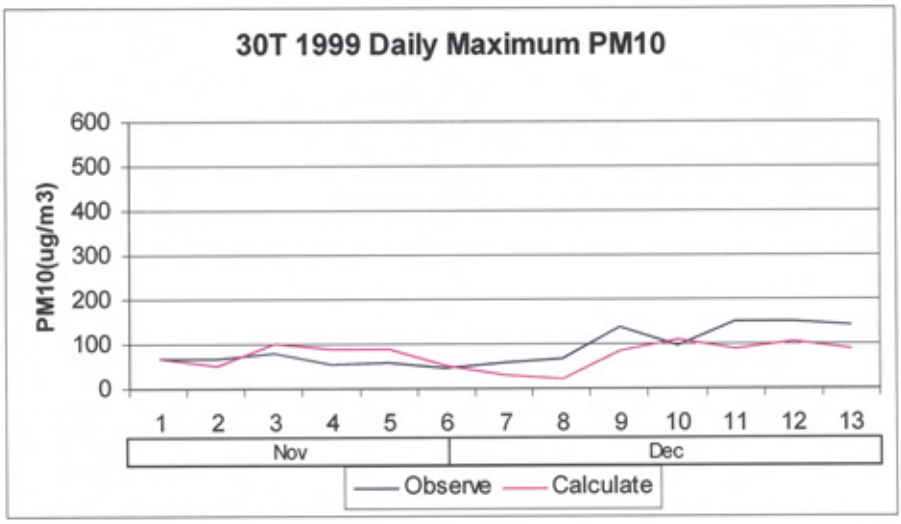
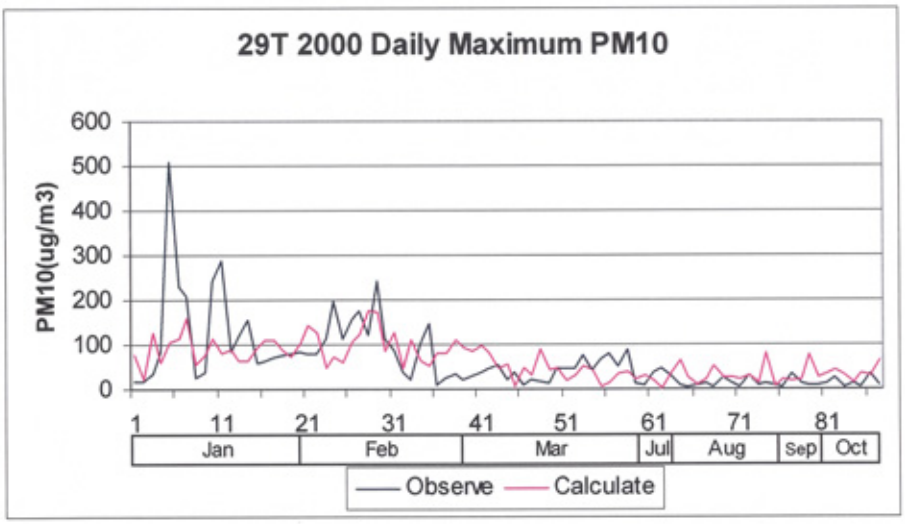


Figure 5.19 Yearly modelled daily maximum PM10 at the 29, 30, 31T station year 1998-2002.



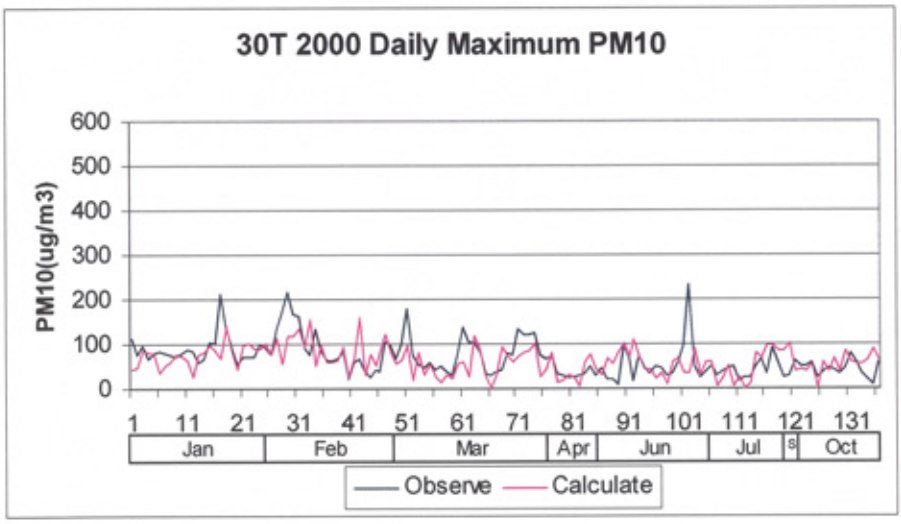
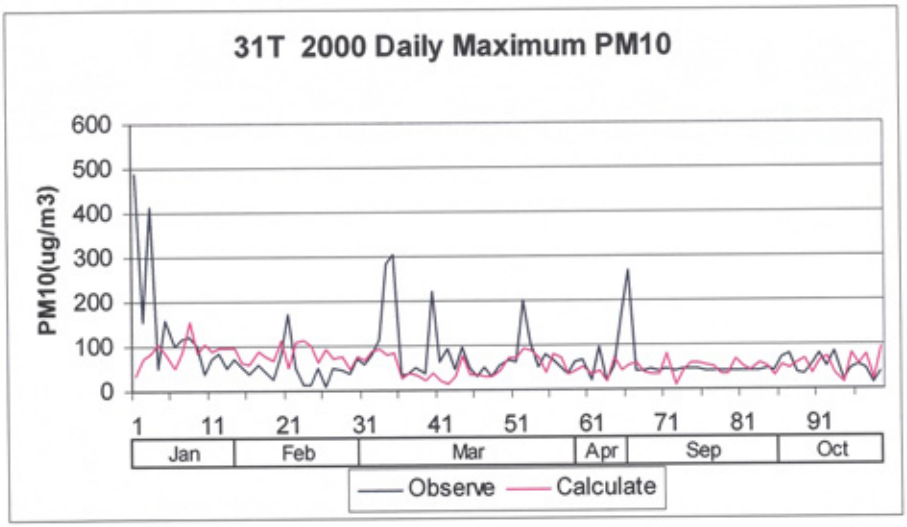


Figure 5.19 Yearly modelled daily maximum PM10 at the 29, 30, 31T station year 1998-2002 (continued).



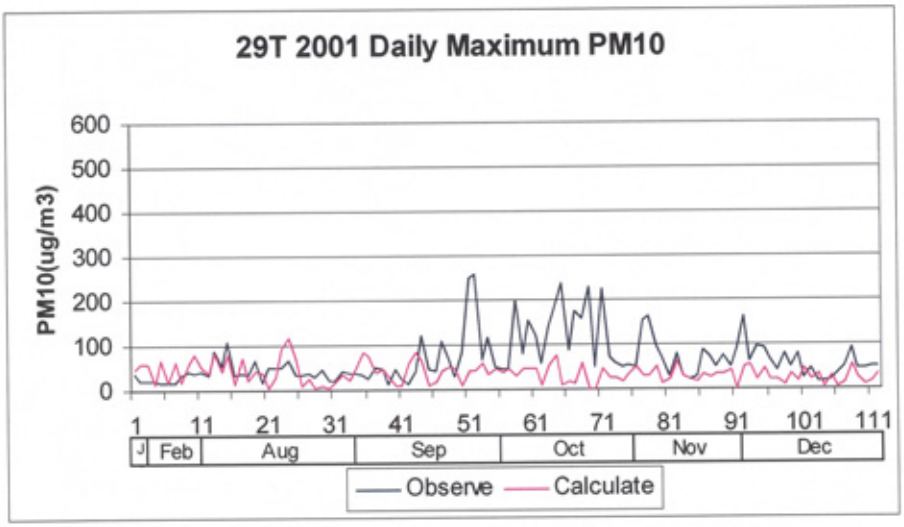
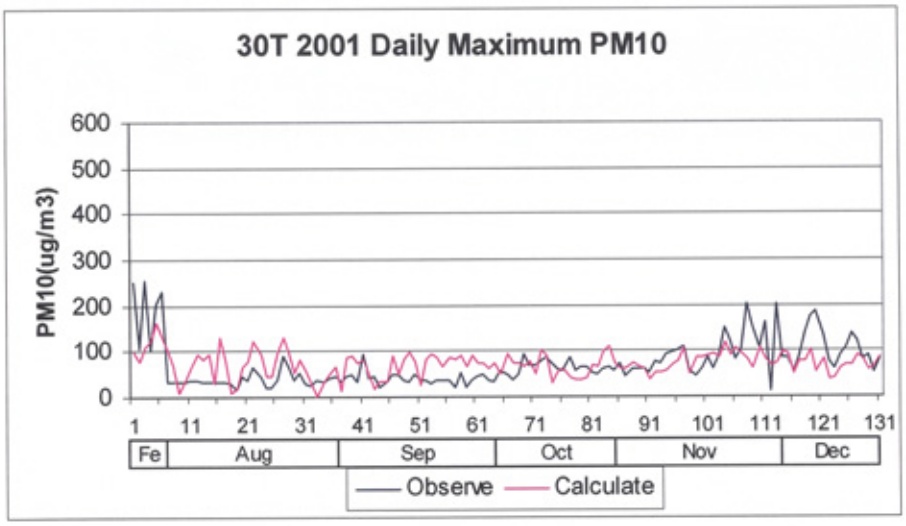


Figure 5.19 Yearly modelled daily maximum PM10 at the 29, 30, 31T station year 1998-2002 (continued).



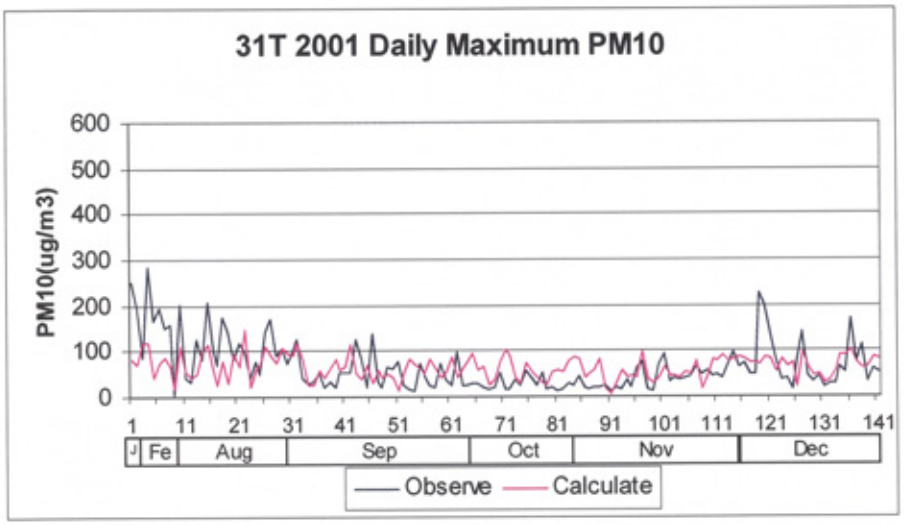
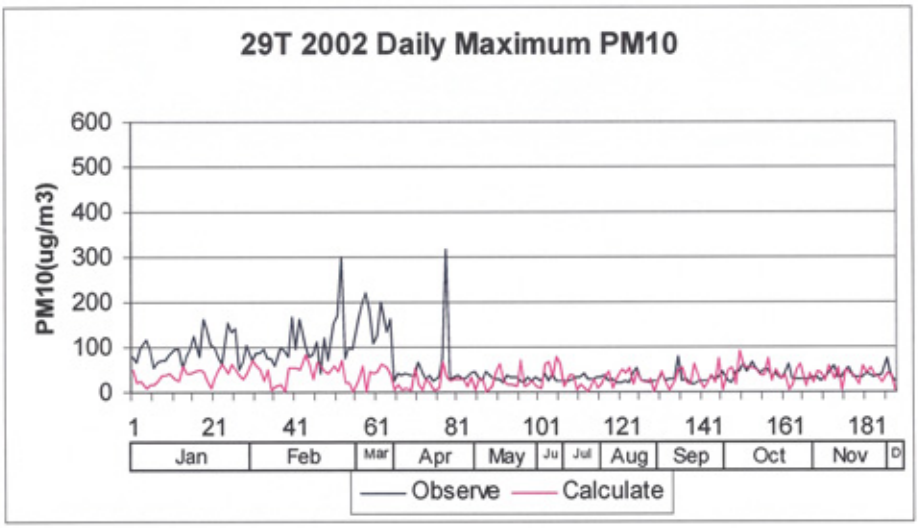


Figure 5.19 Yearly modelled daily maximum PM10 at the 29, 30, 31T station year 1998-2002 (continued).



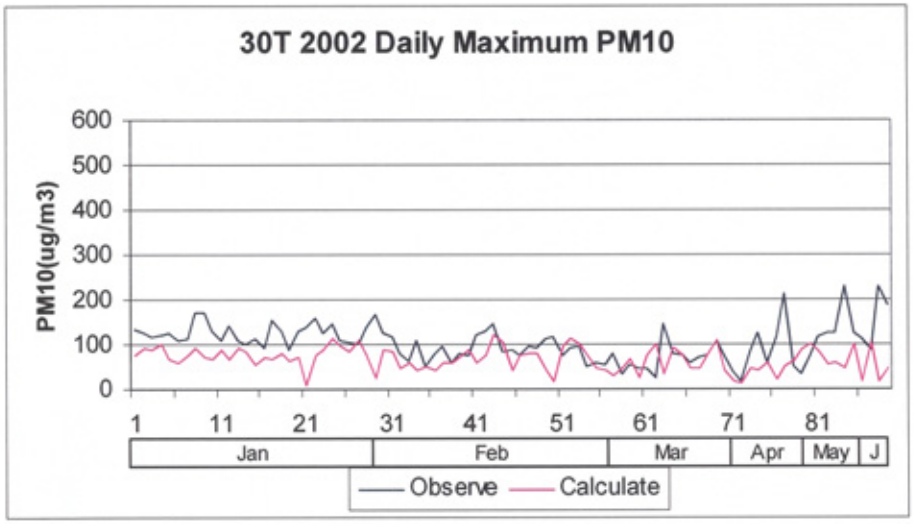


Figure 5.19 Yearly modelled daily maximum PM10 at the 29, 30, 31T station year 1998-2002 (continued).

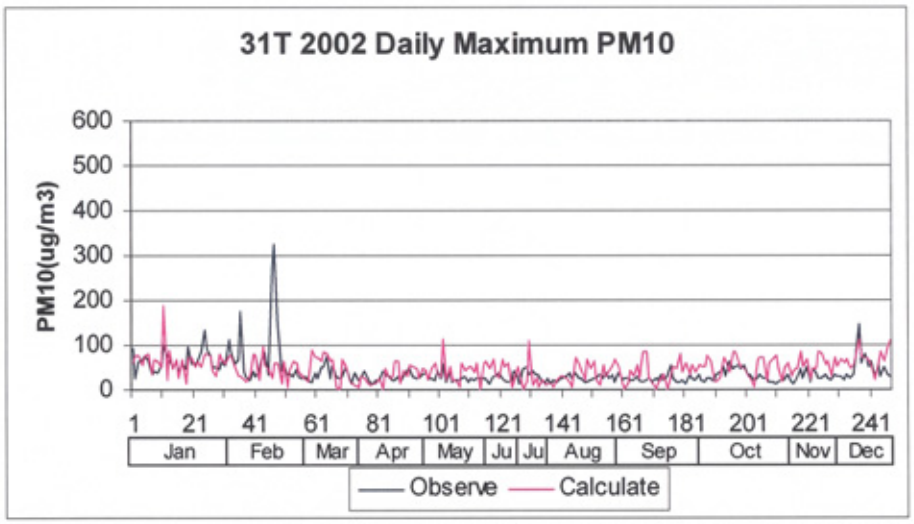


Figure 5.19 Yearly modelled daily maximum PM10 at the 29, 30, 31T station year 1998-2002 (continued).

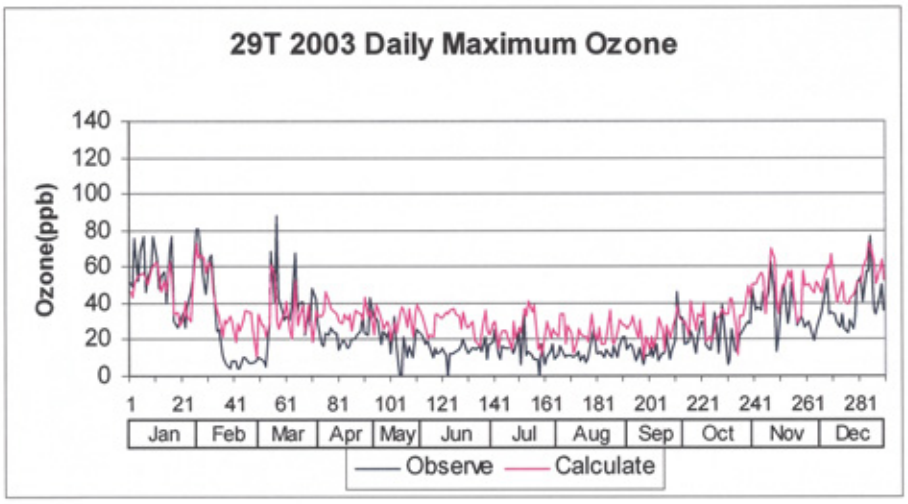
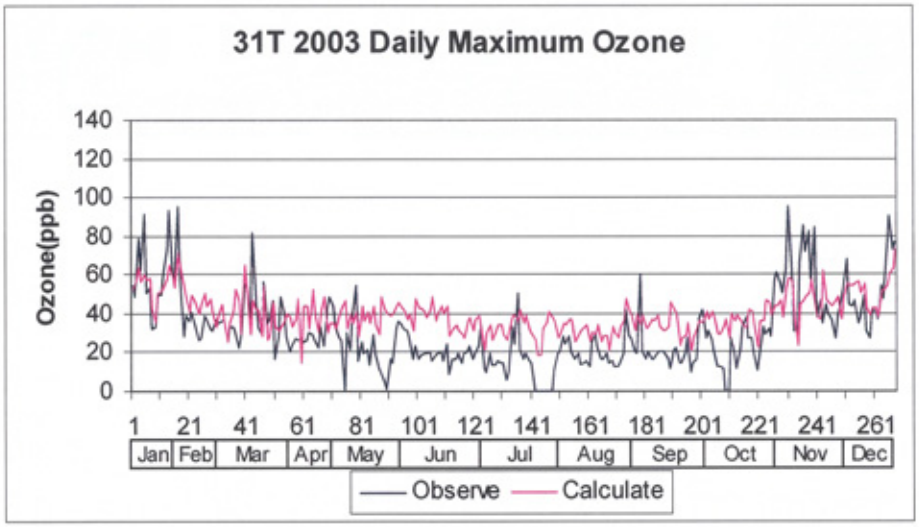


Figure 5.20 Yearly modelled daily maximum Ozone at the 29, 30, 31T station year 2003 – 2004.



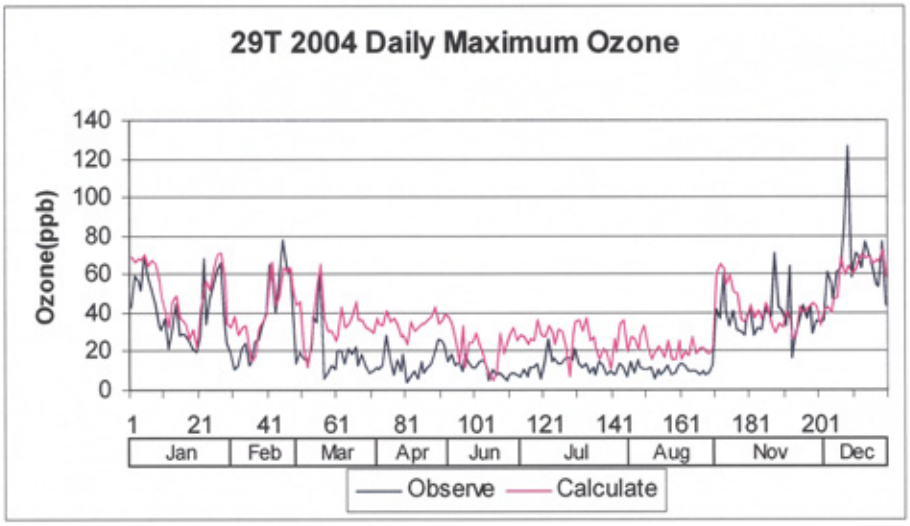


Figure 5.20 Yearly modelled daily maximum Ozone at the 29, 30, 31T station year 2003 – 2004 (continued).

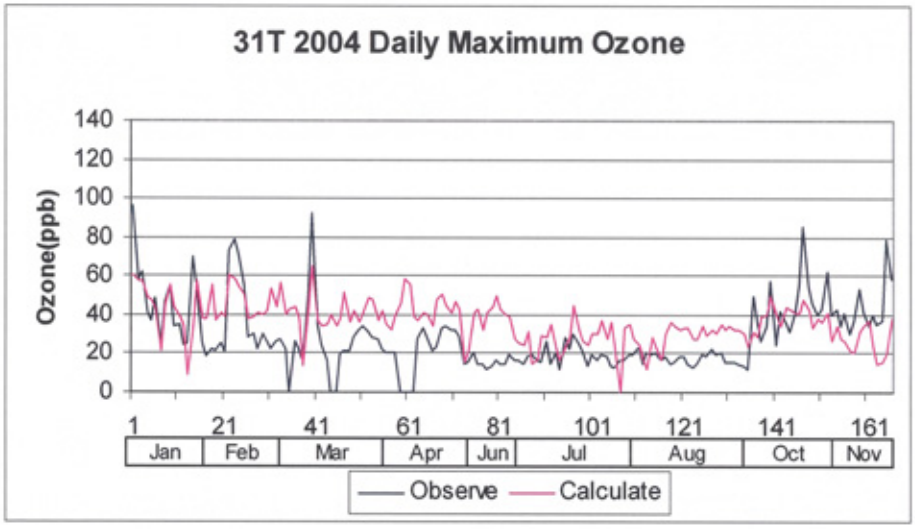
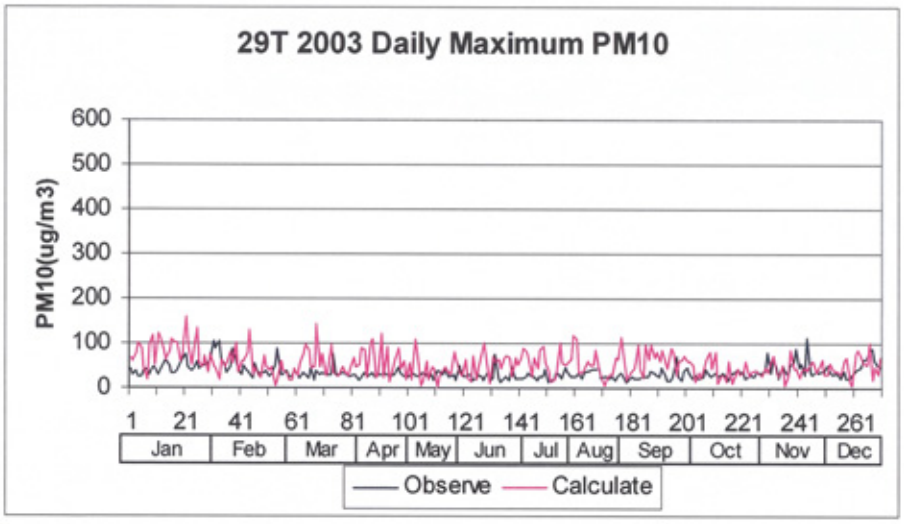


Figure 5.20 Yearly modelled daily maximum Ozone at the 29, 30, 31T station year 2003 – 2004 (continued)
Note: Hydrocarbon data at 30T station is not available since 2003 so daily maximum ozone cannot be calculated.



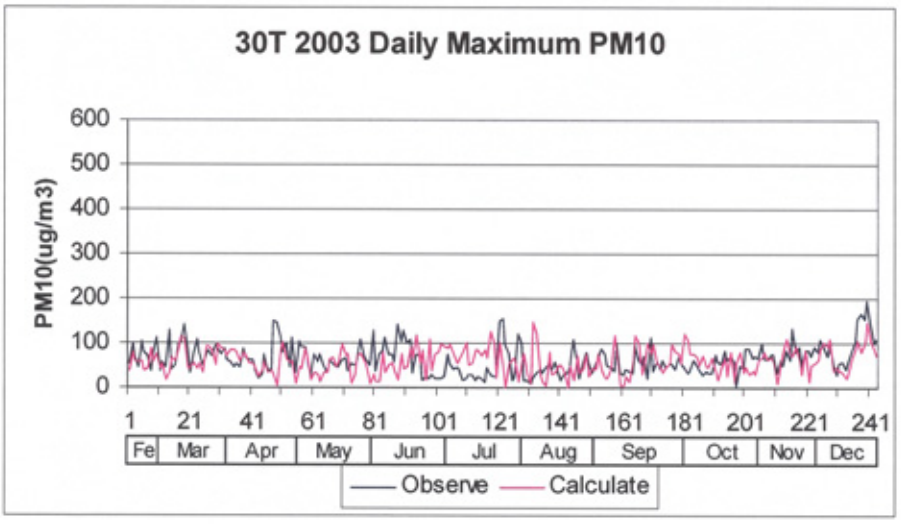
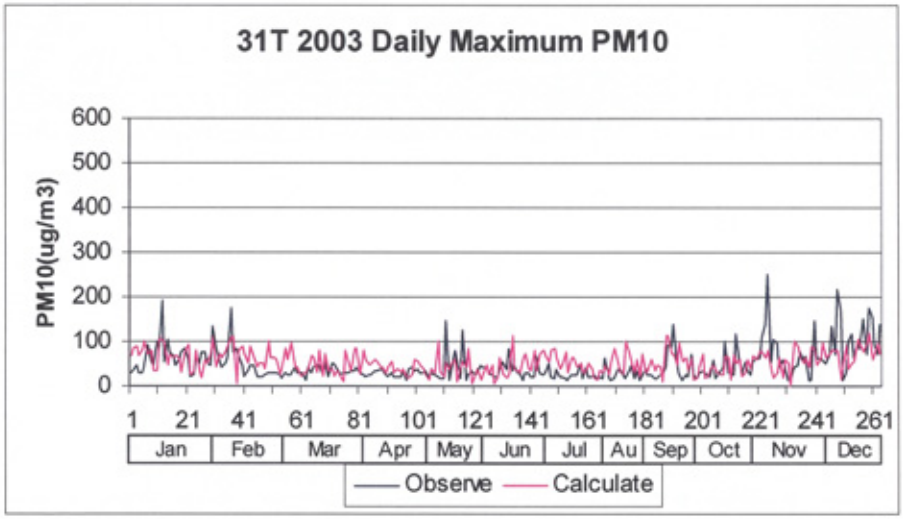


Figure 5.21 Yearly modelled daily maximum PM10 at the 29, 30, 31T station year 2003 – 2004.



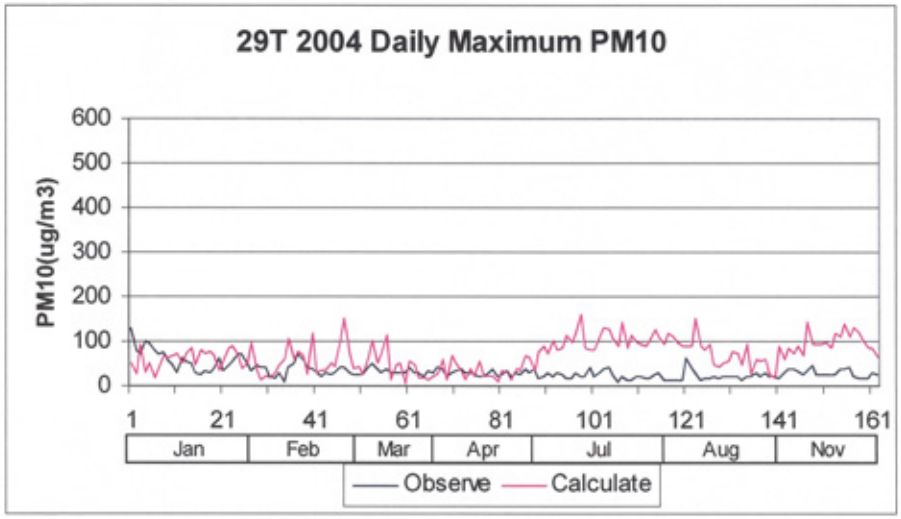
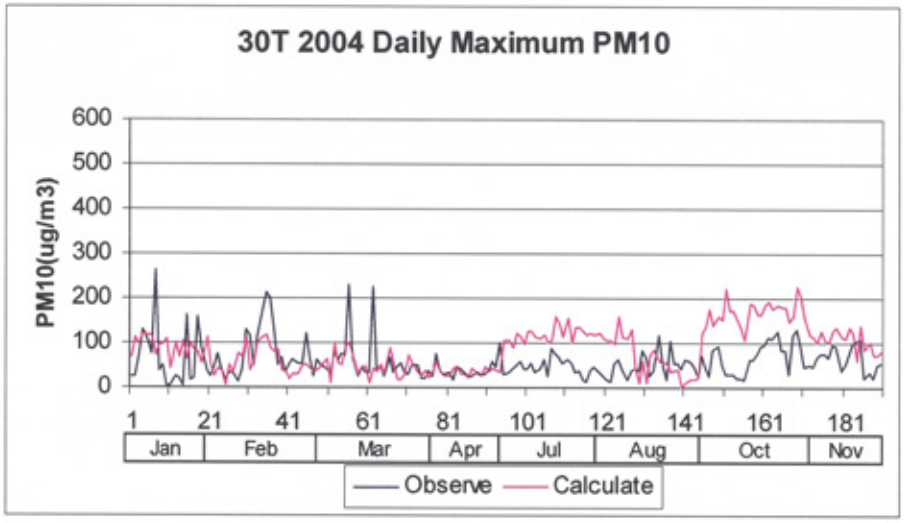


Figure 5.21 Yearly modelled daily maximum PM10 at the 29, 30, 31T station year 2003 – 2004 (continued).



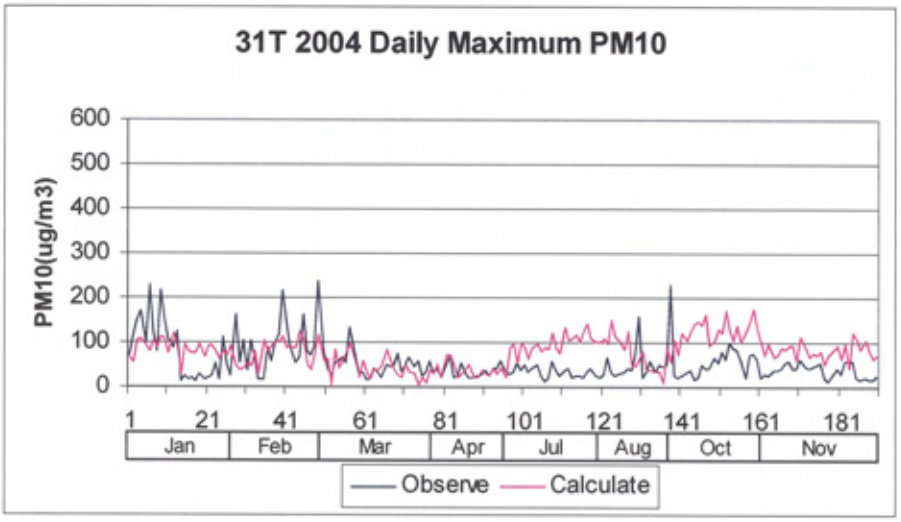


Figure 5.21 Yearly modelled daily maximum PM10 at the 29, 30, 31T station year 2003 – 2004 (continued).

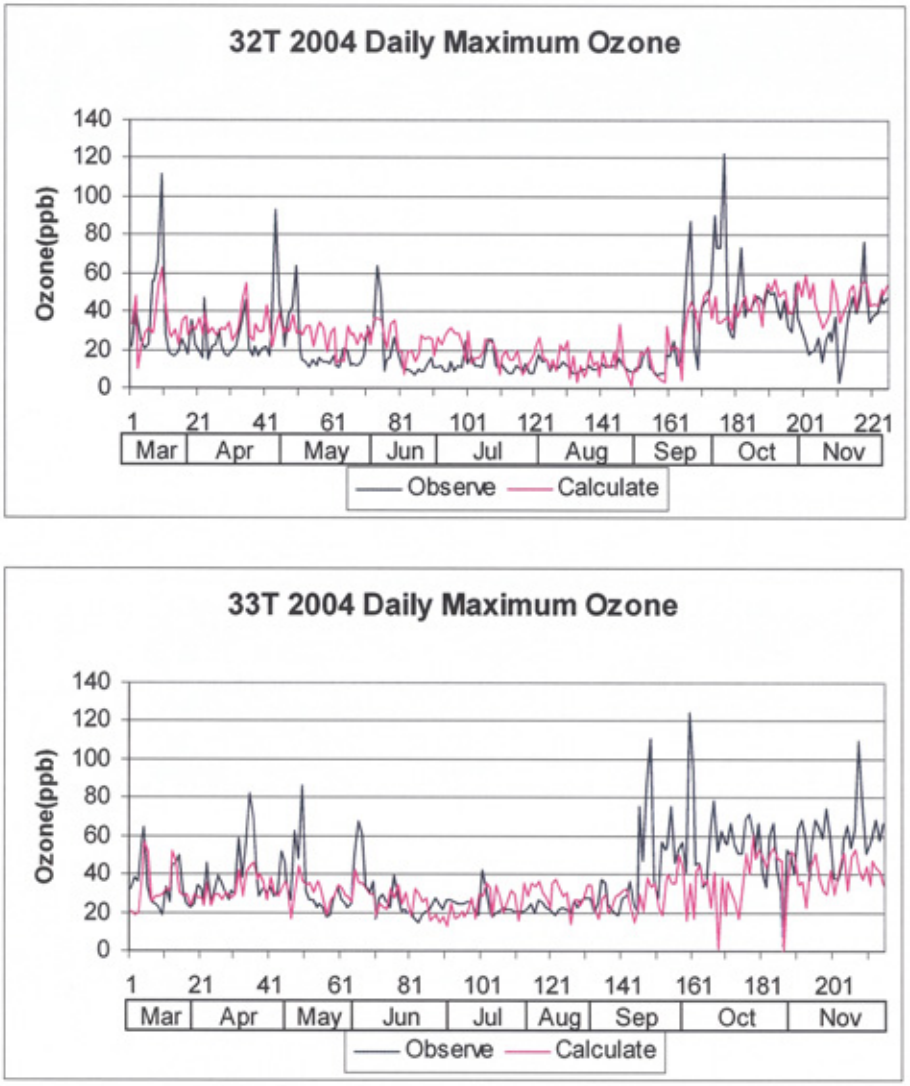


Figure 5.22 Yearly modelled daily maximum Ozone at the 32 and 33T station, Chonburi, year 2004.

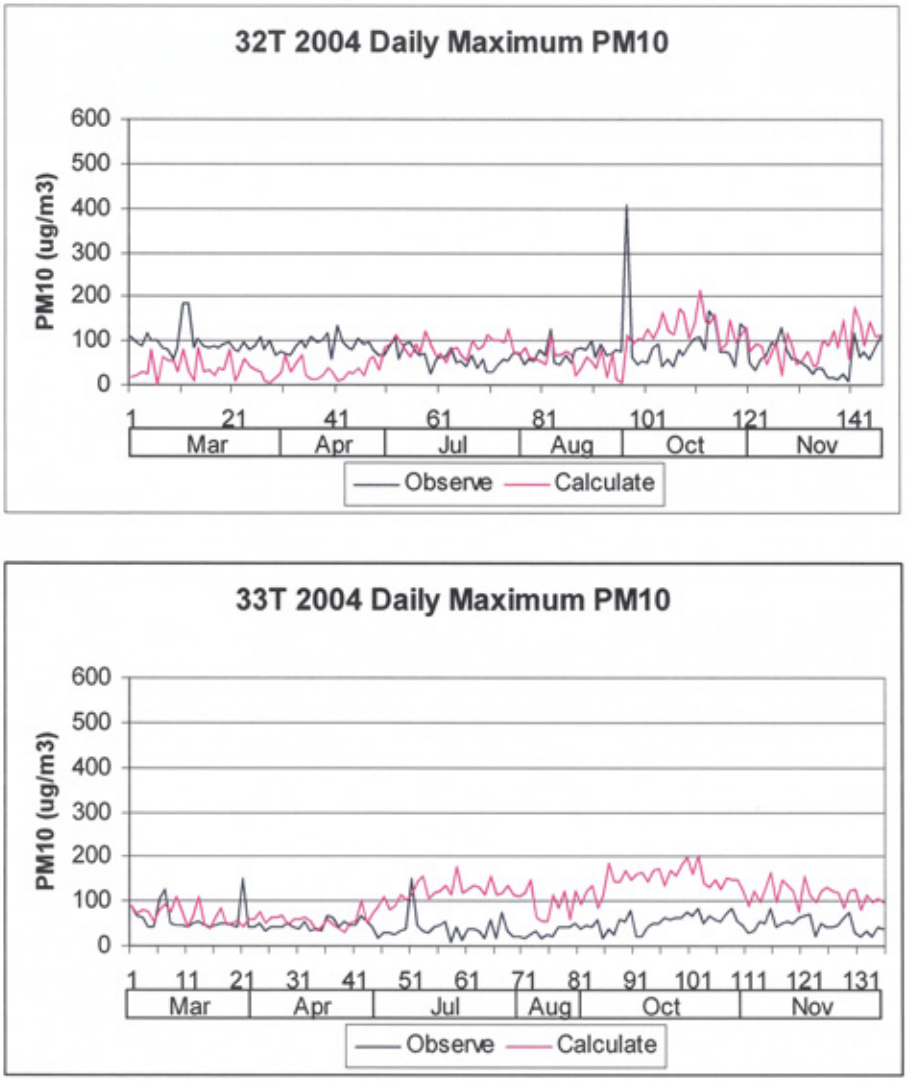


Figure 5.23 Yearly modelled daily maximum PM10 at the 32 and 33T station, Chonburi, year 2004.

5.4 Health Impact Analysis

The time series analysis for Rayong City and for Maptaphut reported on in Chapter 4 showed a clear correlation between pollutant concentrations and hospital admissions, principally due to PM_{10} and to a lesser extent nitric oxide and ozone. For Rayong, the main contributions will be a variety of city-based sources such as traffic, local industry, and the use of fossil fuels to power generators. With the town of Maptaphut and its close proximity to the industrial estate there is the potential that as well as traffic and general background sources, emissions from the industrial estate could be having a significant health impact on the local population.

To try to elucidate any health impacts from the industrial estate, the monitored and modelled air pollution data was used in combination with the AirQ Health Impact Assessment tool published by the World Health Organisation (WHO European Centre for Environment and Health 2004). This program allows the calculation of health impacts from exposure to specific air pollutants over a defined period of time; the program utilises published values on relative risks per 10$\mu g\ m^{-3}$ increment for a number of different health endpoints, for example mortality (cardiovascular and respiratory) and morbidity (hospital admissions). For the present study only respiratory admissions data was available for the two nearby hospitals to the monitoring sites, and so only this health effect was considered.

The health impact analysis was carried out for NO_2, SO_2, PM_{10} and ozone at ambient air quality monitoring stations 29T (Maptaphut) and 30T (Rayong City). Ambient air quality monitoring data from 2004 and 2005 was separately prepared for input into the AirQ program. For NO_2, SO_2 and PM_{10}, average daily values were calculated from the hourly monitoring data; conversions from ppb to were carried out for NO_2, and SO_2 (AirQ requires concentrations in $\mu g\ m^{-3}$). In the case of ozone, the health endpoint is related to the daily maximum 1 hour maximum concentration and so the above procedure was repeated for this parameter. The frequency (days per year) that the average daily pollutant concentrations (or maximum daily 1 hour average in the case of ozone) were within each of the specified concentration ranges was calculated using Excel (10$\mu g\ m^{-3}$ increments up to 200$\mu g\ m^{-3}$, and 50$\mu g\ m^{-3}$ increments thereafter). The calculated frequencies were input into AirQ, which was then used to predict the number of excess respiratory hospital admissions for each pollutant. The relative risk factors, and the predicted number of extra

cases at concentrations above 10μg m^{-3} was calculated using AirQ based on the assumption of populations and admission rates shown in Table 5.3.

Table 5.3 Statistical data for populations and respiratory admissions

	Population		Respiratory admissions (incidence per 100,000 shown in parentheses)	
Population Centre	2004	2005	2004	2005
Maptaphut	40,000	40,000	13,712 (34,200)	21,021 (52,500)
Rayong City	543,887	559,135	47,210 (8,700)	47,856 (8,600)

The respiratory admission data was obtained directly from the two hospitals and it is worth noting that they are unusually high, especially for the Maptaphut hospital. It has been previously noted that in the early years of operation of the Maptaphut industrial estate incidences of respiratory illness were higher than the Thai national average (Jadsri *et al*, 2006), and it may be tempting to assume that the data in Table 5.3 reflects this as a continuing trend. However, incidence rates of 52,500 per 100,000 for Maptaphut hospital in 2005 far exceed the highest rates found by Jadsri et al (2006), which was 23,365 per 100,000 for one area of Maptaphut (rainy season). The average rate for the areas neighbouring the industrial estate in 1996 was 5,649, compared to averages for Rayong Province and Thailand nationally of 4,961 and 3,123 respectively. These latter rates compare reasonably well to the rates observed at the Rayong hospital (8,600) and so we must assume that other factors are contributing to the exceptionally high Maptaphut rates, namely: (a) residents from outside Maptaphut accessing the hospital, thus boosting the rate, and (b) the population figures being inaccurate because of the significant numbers of migrant workers on the industrial estate. Knowledge of the area indicates that both of these are likely to be valid and significant. In the study by Jadsri et al (2006), the area of residence was obtained from the medical records, thus enabling non-Maptaphut residents to be removed from the statistics. Nevertheless, it does seem that the rates in Maptaphut and Rayong City are higher than national figures.

To account for the possibility of non residents boosting the Maptaphut respiratory admissions rate the AirQ programme was run with two separate admission rates: the actual recorded rate and the rate taken from Rayong, which might be considered a more realistic rate. Both sets of data are presented for comparison.

In addition to running AirQ for the ambient monitoring data, the same procedure was repeated for the ADMS modelled data for Maptaphut, thus allowing the health impacts of the Maptaphut contribution to be calculated. All of these data are shown in Table 5.4.

The results of this basic health impact analysis shows that emissions from the Maptaphut industrial estate account for almost all of the NO_2 and SO_2 related respiratory illness and between 10 and 27% of the PM_{10} related admissions. Taking the lower of the incidence rates, 8,700 per 100,000, for the reasons described above, this amounts to between 45 and 60 extra hospital admissions for respiratory illness per year in the Maptaphut area. This figure excludes ozone related respiratory illness, which couldn't be estimated because of inadequacies in the chemistry model of ADMS 3. In addition, there will be admissions for other pollutant related illness, such as cardiovascular effects, and an increased mortality rate. Whilst these numbers might seem high, they represent less than 2% of the total admissions for respiratory illness (a maximum of 60 out of a total of 3440 admissions, based on the lower of 8,700 per 100,000 incidence rate). It does seem odd, therefore, that such a significant correlation was seen in the correlation study reported in Chapter 4; a 2% increment on the baseline is unlikely to be detectable as significant, so clearly there are other factors that are important. Jadsri *et al*, (2006) showed that there were significant clusters of respiratory illness in communities living closer to the Maptaphut industrial state, who were exposed to much higher concentrations of NO_x, SO_2 and PM_{10} from the estate, with incidence rates in some areas almost quadrupling compared to the average for Maptaphut as a whole. A weakness of our basic health impact study is, therefore, that it relates respiratory admissions to monitored data at only one ambient air quality station; there are likely to be communities that are exposed to much higher concentrations, and it is these higher concentrations that are contributing to the excess emissions that in all likelihood represent a much more significant proportion of respiratory admissions than 2%. Whilst there have been improvements in emission standards since 1996, the ambient levels, as discussed in chapter 4 have only slightly improved since that time, most notably for PM_{10}, and so the health issues are likely to be similar.

As mentioned above, thus far the health impact study has concentrated mainly on the most densely populated area of Maptaphut town to the North West of the industrial estate, however there are populated areas all around the industrial estate. Therefore it was decided to make a visual estimate of the potential for some populations around the Maptaphut industrial estate to be exposed to particularly high concentrations of pollutants.

This was done by generating pollution maps for SO_2 and PM_{10} for annual average concentrations of mean hourly data. District information was then superimposed onto the pollution maps. The district information was taken from the recently published study of Jadsri *et al*, (2006).

These plots, shown in Figures 5.24 and 5.25 for SO_2 and PM_{10} respectively, indicate that some populated districts, namely district 9 (Mapchalood, population 1,050), district 22 (Nangpab, population 825), district 23 (Takuanaupradoo, population 1,670) and district 13 (Wasopol, population 1,088) are exposed to significant mean concentrations of SO_2 and PM10. Moreover these areas correspond to respiratory disease clusters identified by Jadsri *et al*, (2006).

Table 5.4: Heath impact analysis for respiratory admissions

Pollutant	Year	Location and year	Population	Total respiratory admissions	Total respiratory admissions per 100,000	Risk Factor	Mean annual concentration of pollutant (µg m-3)	Number of excess cases for ambient monitoring data (upper – lower limits in parentheses).	Number of excess cases due to Maptaphut emissions alone (upper – lower limits in parentheses).	Estimated contribution to excess cases from Maptaphut (%)
PM10	**2004**	**Maptaphut (T29) actual**	40,000	13,712	34,200	1.008	26	182 (110 – 253)	20 (12 – 28)	11
		Maptaphut (T29) adjusted[1]	40,000	-	8,700		26	46 (28 – 102)	5 (3 – 7)	11
		Rayong City (T30)	543,887	47,210	8,700		44	1267 (765 – 1755)	0	0
	2005	**Maptaphut (T29) actual**	40,000	- 21,021	52,500	1.008	16	69 (41 – 96)	18 (11 – 26)	26
		Maptaphut (T29) adjusted	40,000	-	8,600		16	17 (10 – 24)	3 (2 – 4)	18
		Rayong City (T30)	559,135	47,856	8,600		44	1268 (769 – 1757)	0	0
NO2	**2004**	**Maptaphut (T29) actual**	40,000	13,712	34,200	1.0038[2]	22	63 (0 – 197)	74 (0 – 230)[3]	100
		Maptaphut (T29) adjusted	40,000	-	8,700		22	16 (0 – 51)	19 (0 – 59)	100
		Rayong City (T30)	543,887	47,210	8,700		22	232 (0 – 724)	0	0
	2005	**Maptaphut (T29) actual**	40,000	- 21,021	52,500	1.0038	18	97 (0 – 302)	87 (0 – 270)	90
		Maptaphut (T29) adjusted	40,000	-	8,600		18	16 (0 – 50)	14 (0 – 44)	87
		Rayong City (T30)	559,135	47,856	8,600		20	193 (0 – 603)	0	0
SO2	**2004**	**Maptaphut (T29) actual**	40,000	13,712	34,200	1.004[2]	22	72 (18 – 161)	73 (18 – 163)	100
		Maptaphut (T29) adjusted	40,000	-	8,700		22	18 (5 – 41)	19 (5 – 41)	100
		Rayong City (T30)	543,887	47,210	8,700		9	44 (11 – 99)	0	0
	2005	**Maptaphut (T29) actual**	40,000	- 21,021	52,500	1.004	16	92 (23 – 205)	92 (23 – 205)	100
		Maptaphut (T29) adjusted	40,000	-	8,600		16	15 (4 – 34)	15 (4 – 34)	100
		Rayong City (T30)	559,135	47,856	8,600		7	21 (5 – 48)	0	0
Ozone	**2004**	**Maptaphut (T29) actual**	40,000	13,712	34,200	1.0062[2]	30	380 (186 – 567)	-	-
		Maptaphut (T29) adjusted	40,000	-	8,700		30	95 (47 – 143)	-	-
		Rayong City (T30)	543,887	47,210	8,700		34	1455 (716 – 2172)	-	-
	2005	**Maptaphut (T29) actual**	40,000	- 21,021	52,500	1.0062	38	653 (321 – 974)	-	-
		Maptaphut (T29) adjusted	40,000	-	8,600		38	107 (53 – 160)	-	-
		Rayong City (T30)	559,135	47,856	8,600		34	1503 (739 – 2240)	-	-

[1] Assume an admissions rate equivalent to that of Rayong hospital
[2] No specific risk factor available for total respiratory hospital admissions, so risk factor used for >65 years (represents 87% of respiratory illness admissions)
[3] ADMS predicts NO_x concentrations; this comparison was made on the basis of assuming 100% conversion to NO_2, which is likely to overestimate the impact.

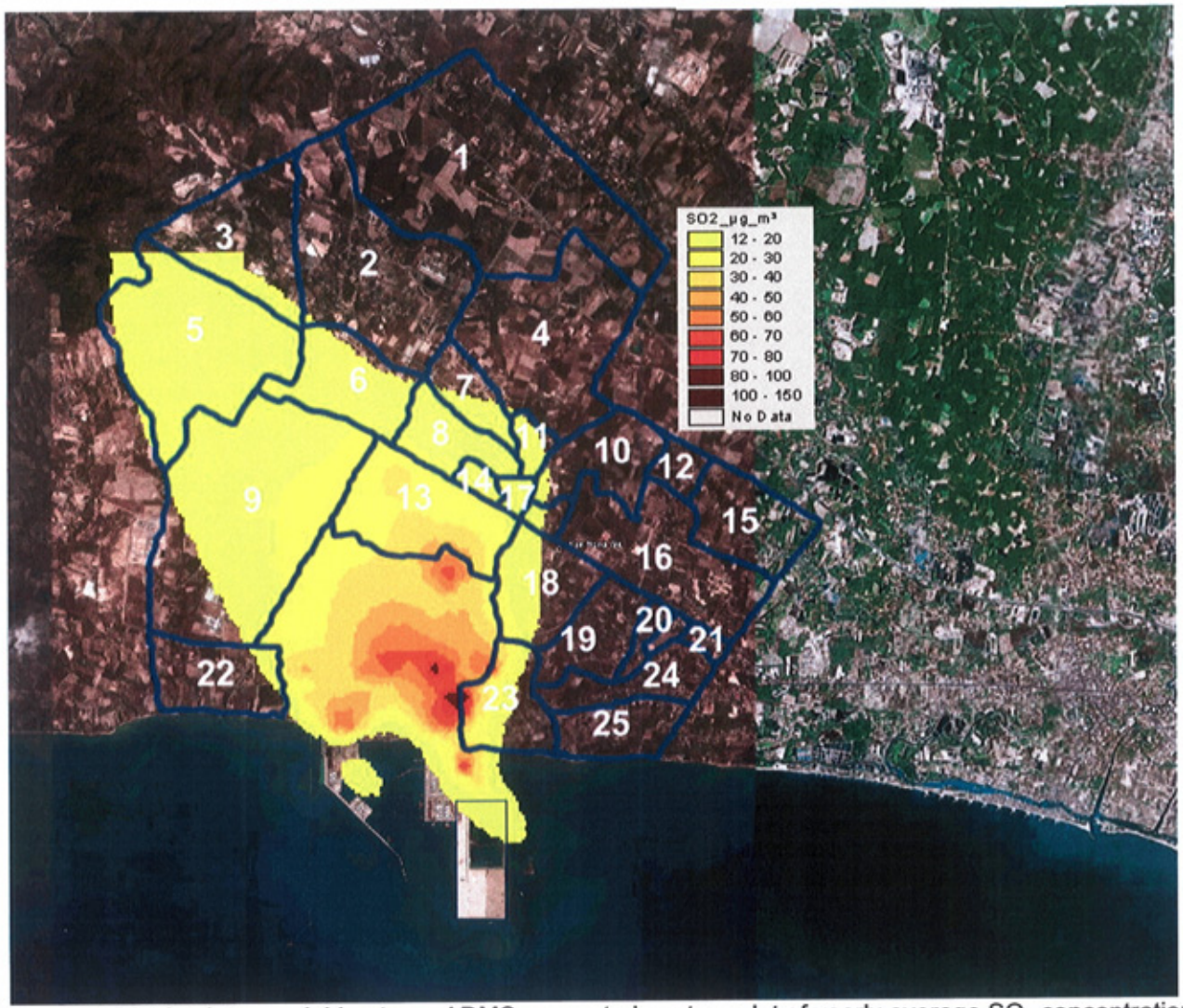


Figure 5.24 District data overlaid onto an ADMS generated contour plot of yearly average SO_2 concentrations

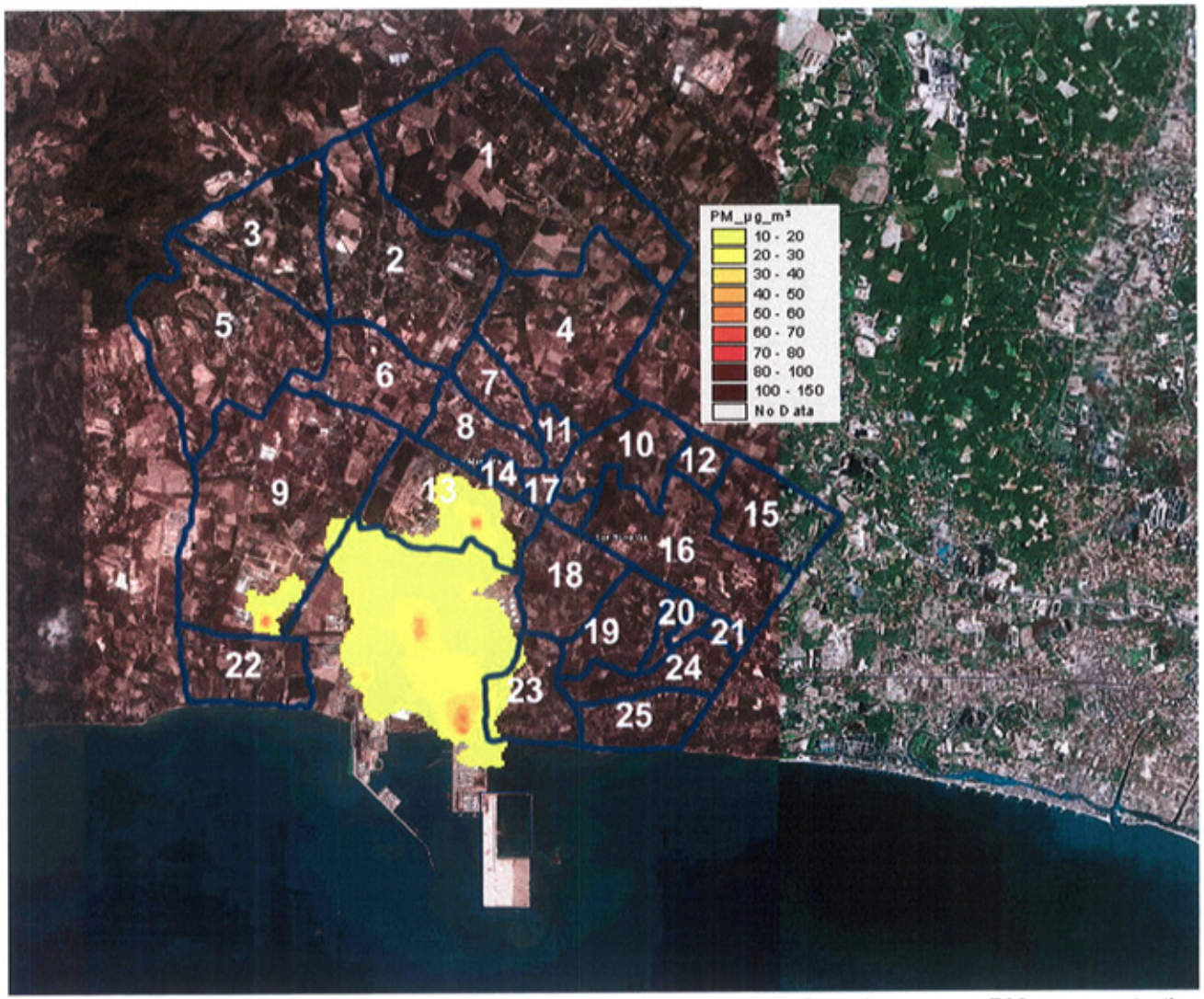


Figure 5.25 District data overlaid onto an ADMS generated contour plot of yearly average PM_{10} concentrations

Chapter 6

Conclusions and recommendations

6.1 Conclusions

6.1.1 Analysis of meteorological and ambient air quality data for the Maptaphut / Rayong region

The wind rose analysis shows that emissions from the industrial estate will be blown predominantly towards population centres during January to September due to the south-western wind, whereas during October to December the predominant north-eastern wind will tend to disperse the emissions into the Gulf of Thailand. In addition, dispersion and build-up of pollutants in the Maptaphut Industrial Estate can be strongly affected by the sea-land breeze circulation. Continuous coastal fumigation may occur and result in high concentrations of pollutants in the inland area.

The analysis has shown that hourly O_3 and 24-hour average PM_{10} are likely to be the air pollutants of concern for the Maptaphut Industrial Estate area during the dry season. The maximum frequency for exceeding the NAAQS for O_3 is 0.7% and for PM_{10} is 13.4% with the respective maximum values of 128 ppb and 334 μg m^{-3}. Air pollution in the wet season is low. The pollution parameters such as 1-h CO, SO_2, and NO_2 are well within NAAQS all year round and much lower than in Bangkok. There is no clear trend of air pollution in the Maptaphut Industrial Estate during the considered ten-year period (1998-2007), though there is a slight increase in average PM_{10} and O_3 at two stations from 1999 to 2001.

Diurnal variations clearly show that during the wet season and the transitional period from May to October inclusively, the Maptaphut Industrial Estate area is not likely to have an air quality problem. Two air pollution peaks in the diurnal status corresponding to the morning and evening rush hours can be distinguished, while midday concentration levels drop significantly by convective mixing of the boundary layer. Hence, the significance of traffic emission source can be seen by high peaks of NO_x, PM_{10}, CO and NMHC concentrations in the morning and evening rush hours.

A simple directional analysis approach for pollutants and sources has been developed in order to ascertain the main sources impacting on particular monitoring points. The analysis

clearly indicated that the main source for sulphur dioxide was the industrial estate, as expected, but also showed the influence of road and city based sources for the other major pollutants.

6.1.2 Health impacts of air pollution in Maptaphut

Admission trends for Maptaphut and Rayong hospitals, are almost identical, suggesting that the same factors affecting admissions in Rayong are also prevalent in Maptaphut, perhaps indicating that emissions from the industrial estate are not a significant factor (since they have little influence on air quality in Rayong itself, which is 30km from the industrial estate). Multiple linear regression carried out for air pollutants and respiratory disease outpatient data for Maptaphut hospital suggests that exposure to ozone and NO for longer period up to 4 weeks could result in higher admission rates. In addition, temperature and % RH are significant factors.

High rate of respiratory diseases in the studied area suggests a causal link to air pollution during dry season. Further research would be necessary to reveal the health effects from air pollution in Rayong. Nevertheless, results of a basic health impact analysis using the ADMS modelled concentrations and the WHO AirQ tool, along with default risk factors, shows that emissions from the Maptaphut industrial estate account for almost all of the NO_2 and SO_2 related respiratory illness and between 10 and 27% of the PM_{10} related admissions. This amounts to between 45 and 60 extra hospital admissions for respiratory illness per year in the Maptaphut area (excluding ozone related respiratory illness, which couldn't be estimated because of inadequacies in the chemistry model of ADMS 3). Whilst these numbers might seem high, they represent less than 2% of the total admissions for respiratory illness (a maximum of 60 out of a total of 3440 admissions, based on the lower of 8,700 per 100,000 incidence rate). Jadsri *et al*, (2006) showed that there were significant clusters of respiratory illness in communities living closer to the Maptaphut industrial state, who were exposed to much higher concentrations of NO_x, SO_2 and PM_{10} from the estate, with incidence rates in some areas almost quadrupling compared to the average for Maptaphut as a whole. A weakness of our basic health impact study is, therefore, that it relates respiratory admissions to monitored data at only one ambient air quality station; there are likely to be communities that are exposed to much higher concentrations. Further research would need to be carried out on area-specific health impacts around Maptaphut. The use of mobile monitoring stations, or the installation of

additional stations in those areas likely to be subjected to the highest concentrations of pollutants is advised (29T and 31T are located in relatively unaffected areas).

6.1.3 Modelling studies

The benefits of statistical models are that they are relatively cheap to use, once developed, and that they can be associated with a high degree of accuracy. In addition, the models do not require the sources to be characterised, merely that pollutant emissions are constant, or that in cases of seasonal or daily variability that this is taken into account. For the models developed in this study, there is better agreement between the observed and predicted concentrations for O_3 than for PM_{10}. The models underestimated the peak values of both pollutants, especially for PM_{10}. Statistical models are normally not capable of predicting peak values and the models obtained in this study are not exceptional. An improvement of the statistical prediction can be made if the meteorology was first classified into clusters with homogeneous conditions. The coefficient of determination (R^2) for the O_3 model was 48%, which is higher than the PM_{10} model. The regression model is strongly significant at 0.01 significance level. O_3 was shown to positively correlate with its precursors of NMHC and NO_2 and negatively correlate with NO, which should be expected from the O_3 chemistry. When the models were applied to predict concentrations for years beyond the series for which they were developed, the average percentage errors in the predictions for O_3 and PM_{10} were 21% and 38%, respectively. The models were not applicable for regions other than he none for which they were developed.

For PM_{10}, it is likely that the poor performance of the statistical model is due to the relatively small set of PM_{10} monitoring data. For more accuracy, future research should determine the prediction equation using a much larger set of monitoring data.
ADMS has been shown to predict sulphur dioxide concentrations with a reasonable degree of accuracy; both the magnitude of the peaks and the general trends are reasonably characterised. This indicates that the meteorology model used in ADMS, and the treatment of the boundary layer and Monin-Obukhov lengths is applicable to South East Asian climatic conditions.

For PM_{10} and NO_x, where there are significant other sources apart from the industrial estate, as indicated by the directional analysis work, the characterisation was less good, with some significant under predictions of PM_{10} in particular.

Future work for ADMS modelling should attempt to characterise more fully the NO_x and PM_{10} sources. In addition, some of the more advanced ADMS features should be more fully explored for this setting, i.e. chemistry, precipitation, terrain and coastline.
The predicative capability of ISC for SO_2, NO_x and PM_{10} is generally very poor. There was some coincidence of peaks, but generally the peaks were over predicted for SO_2 and NO_2 and under predicted for PM_{10}. Some of the factors contributing to the relatively poor NO_x and PM_{10} predictions when using ADMS may also apply here, though ISC3 incorporates a much simpler model of vertical turbulence within the atmospheric boundary layer compared with ADMS and AEROMOD both of which use a bimodal distribution of vertical turbulence within the atmosphere; clearly is an important factor. These findings are in accord with literature evaluation studies.

For PanEIA, which is a this computational fluid dynamics (CFD) dispersion model, the model was able to predict general trends and peaks, particularly for SO_2; however there was a tendency to significantly overestimate the daily average concentrations. In addition, run times are significantly greater than those for either ADMS or ISC, possibly by a factor of 10.

6.2 Recommendations

The following recommendations, arising from this research, are addressed to regulatory authorities in Thailand, including the Thai Pollution Control Department, the Thai Ministry of Public Health and the Industrial Estate Authority of Thailand.

This research has shown that advanced Gaussian atmospheric dispersion models such as ADMS 3 are capable of predicting ambient concentrations of air pollutants from well characterised sources. This model gives far superior predictive capability compared to the currently used ISC3 model. It is recommended that ADMS (version 4, now available) be adopted for regulatory purposes, both by the Thai Pollution Control Department and the Industrial Estates Authority of Thailand. Alternatively, should continuation with US EPA methodology be preferred, it is recommended that model evaluation be carried out on the AEROMOD advanced Gaussian dispersion model; this model has a very similar treatment of the vertical atmosphere profile to ADMS and has been found to give comparable results in inter-comparison studies.

It is further recommended that having shown the ADMS algorithms and atmospheric structure models are appropriate to meteorological conditions encountered in Thailand, including in coastal regions, the model (ADMS-Roads or ADMS Urban) be evaluated for modelling traffic related pollution in cities.

This research has shown that statistical approaches to predicting PM_{10} and ozone concentrations may have some application, however such models should be developed and validated in conjunction with specific data sets collected from mobile meteorological and air quality monitoring stations. It is recommended that further research be carried out on these approaches.

The health impact analysis work carried out as part of this project has shown the importance of more widespread ambient air quality monitoring around the Maptaphut industrial estate. There is a suggestion that respiratory admission rates may be much higher in the Maptaphut / Rayong areas compared to the national rates, and it is important that pollutant levels are more adequately described. The contour plots that we have produced for PM_{10} and SO_2 using ADMS show that ambient air quality in populated areas adjacent to the Maptaphut estate is not adequately characterised by the locations of the current monitoring stations. It is recommended that either (a) one of the existing ambient air quality monitoring stations, for example T31, be re-located to a district where higher pollutant levels are encountered, such as District 23 in the South West (Takuanaupradoo) (b) new ambient air quality monitoring stations be installed in areas such as District 23, or (c) mobile air quality monitoring stations are used routinely to collect air quality data at several points around the industrial estate. The Thai Pollution Control Department do have mobile monitoring stations, so recommendation (c) may present the most cost effective solution.

6.3 Further Work

This research has highlighted a number of additional studies that should be carried out. Firstly, it is clear pollution sources in the Maptaphut / Rayong area are inadequately characterised, particularly for PM_{10} emissions, but also for NO_2; sulphur dioxide appears from modelling studies to be reasonably well characterised. Accurate pollution inventories are essential for both modelling studies and health impact analyses. There are industrial estates and power stations to the East of Rayong City, together with road sources and

wind blown dust sources that need to be characterised; our directional analysis studies have indicated where these main sources lie.

Secondly, the use of the chemistry and coastal modules of ADMS should be further investigated in order to build a more accurate model, including atmospheric chemistry. Ozone concentration was poorly predicted by the model, and this should be further explored.

Thirdly, other dispersion models should be evaluated for this study area, particularly AEROMOD, which is a US EPA advanced Gaussian model; it has a similar approach to ADMS in modelling the atmosphere, however because it is part of the same family as ISC3 it may be preferred to ADMS. Intercomparison studies have found very similar performance for both ADMS and AEROMOD.

Fourthly, further work needs to be carried out on statistical techniques for predicting ambient pollutant concentrations, preferably in conjunction with additional monitoring data from mobile monitoring stations. Other statistical techniques should be used, for example ARIMA, which has been widely used in literature studies.

Finally, additional work needs to be carried out on the health impact analysis. There are additional data requirements for ambient air quality monitoring data, particularly in the districts that are exposed to higher pollution concentrations; this could be addressed by use of mobile monitoring station, as mentioned in the previous section. In addition, more detailed hospital admission data is required, including: age, sex, district of residence; none of this data was available for the present study, and so it was not possible to screen out non-Maptaphut residents from the admissions data. Daily rather than weekly data should also be obtained.

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Appendix A

100m Mast Meteorological Data for Rayong and Bangkok

A.1 100 m Meteorological Tower Wind Data

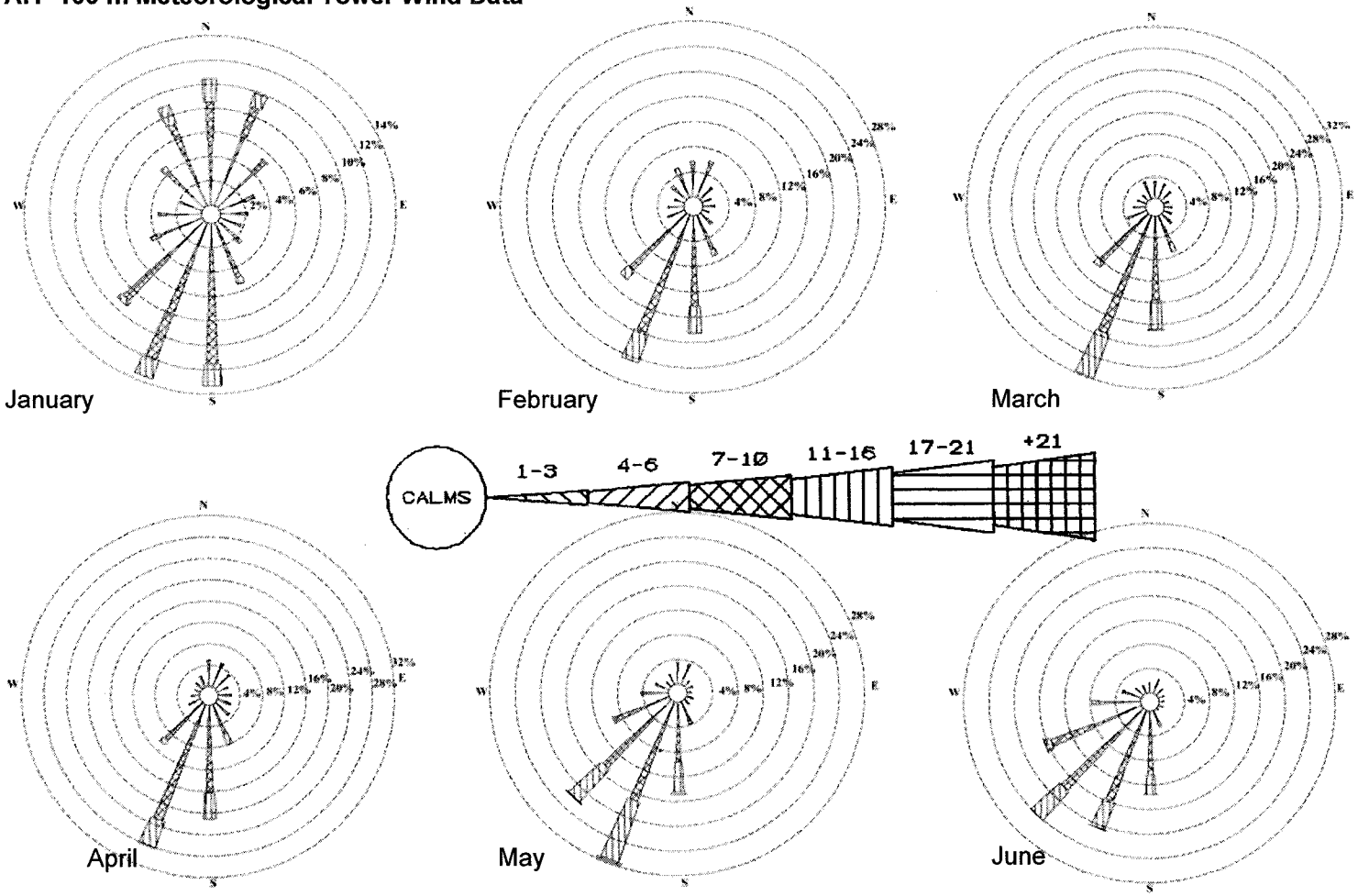


Figure A1. Wind roses at 50 m above ground based on 1998-2002 data at 100 m tall meteorological mast.
Note: Direction is from which the wind blows.

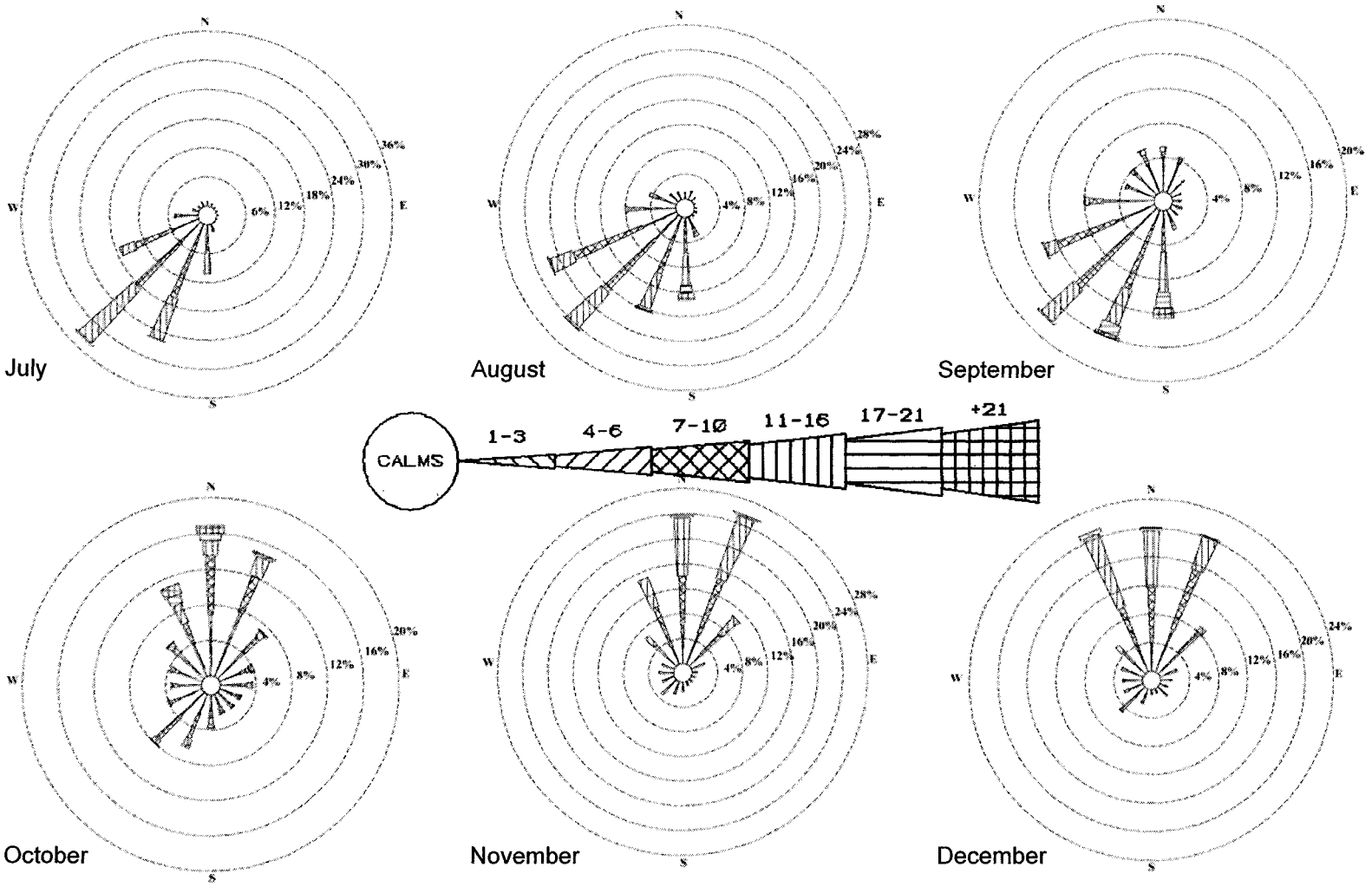


Figure A1. Wind roses at 50 m above ground based on 1998-2002 data at 100 m tall meteorological mast (continued).
Note: Direction is from which the wind blows. Wind Speed in knots

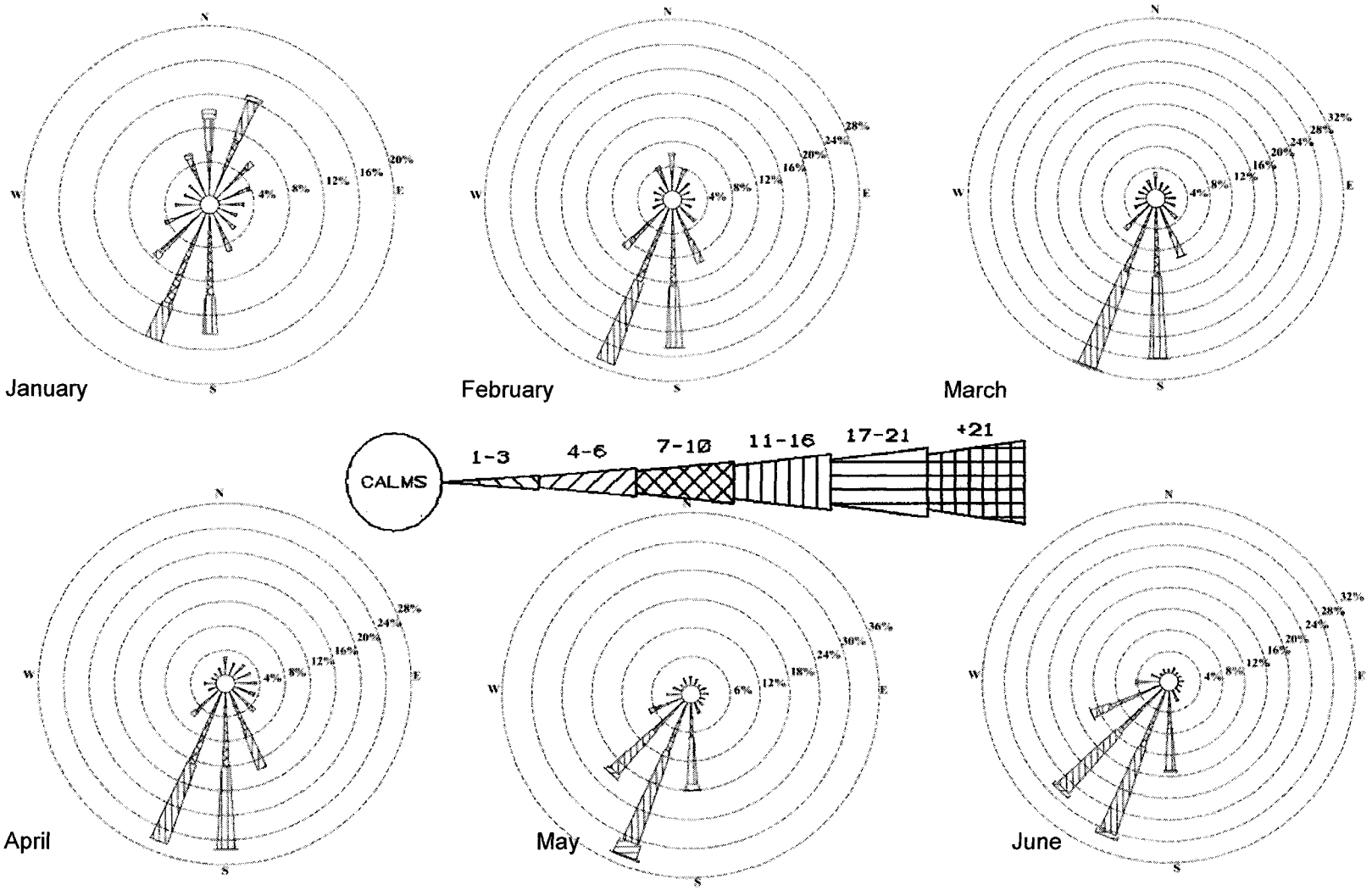


Figure A2. Wind roses at 100 m above ground based on 1998-2002 data at 100 m tall meteorological mast.
Note: Direction is from which wind blows. Wind Speed in knots

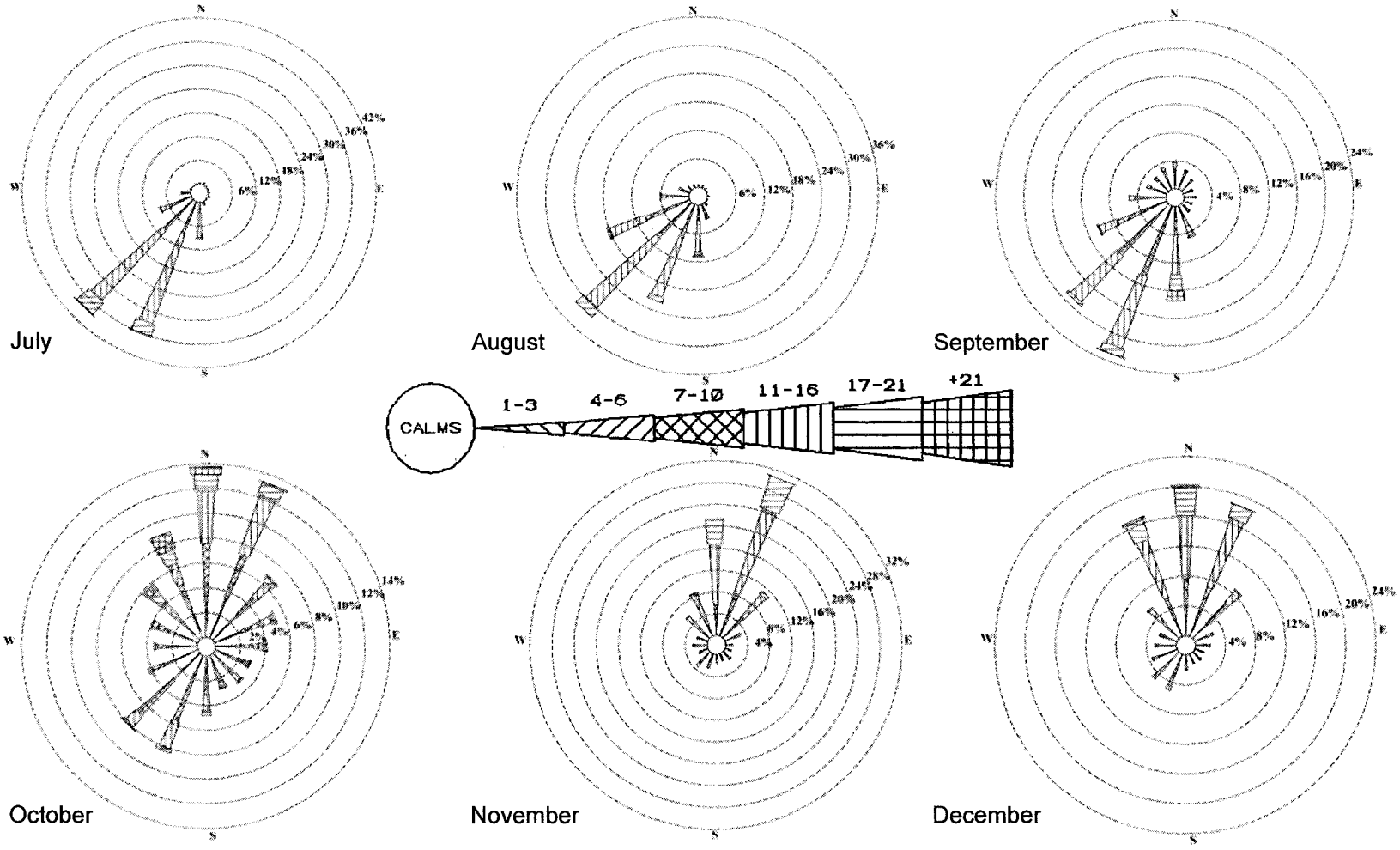


Figure A2. Wind roses at 100 m above ground based on 1998-2002 data at 100 m tall meteorological mast (continued). Note: Direction is from which wind blows. Wind Speed in knots

A.2 Bangkok Wind Data

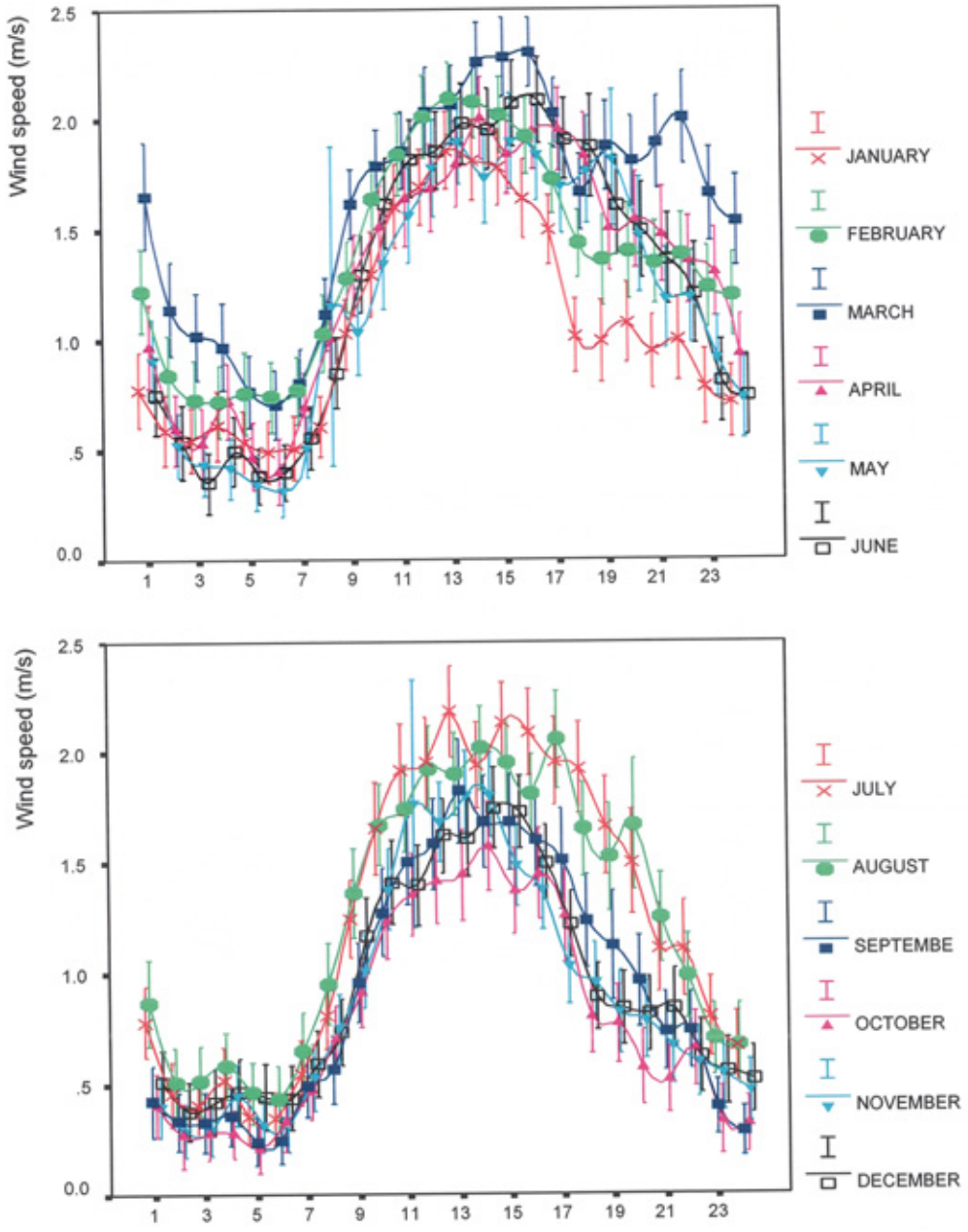


Figure A3. Average wind speed observed at Bangkok Metropolis Station. Vertical bars represent two standard errors of mean values.

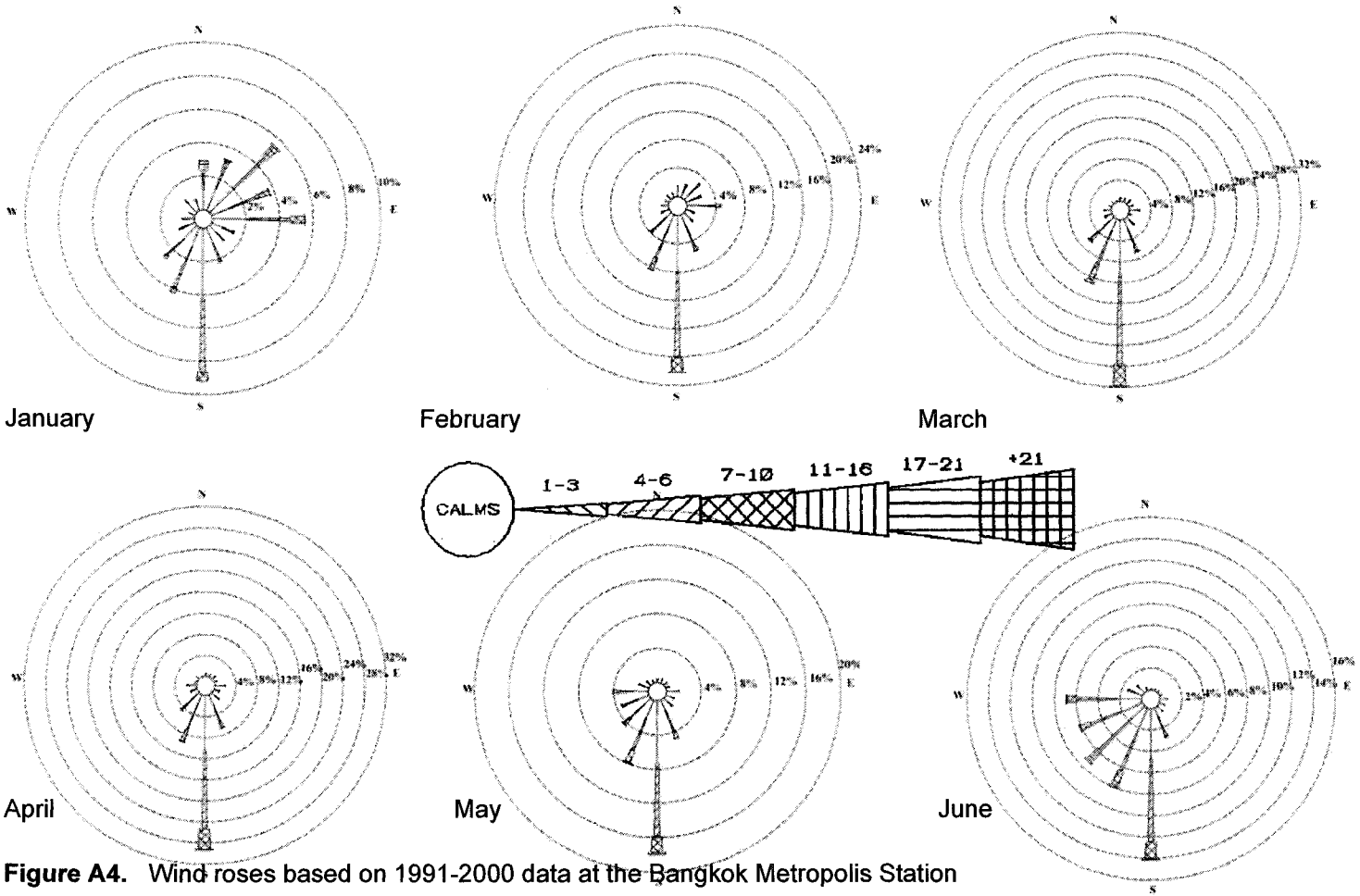


Figure A4. Wind roses based on 1991-2000 data at the Bangkok Metropolis Station

Note: Direction is from which the wind blows. Height of wind vane is 33 meters above ground. Wind Speed in knots

Calm wind frequency is 54%, 39%, 30%, 37%, 49%, 48%, 46%, 48%, 60%, 61%, 53%, and 52% for January, February, March, April, May, June, July, August, September, October, November, and December, respectively.

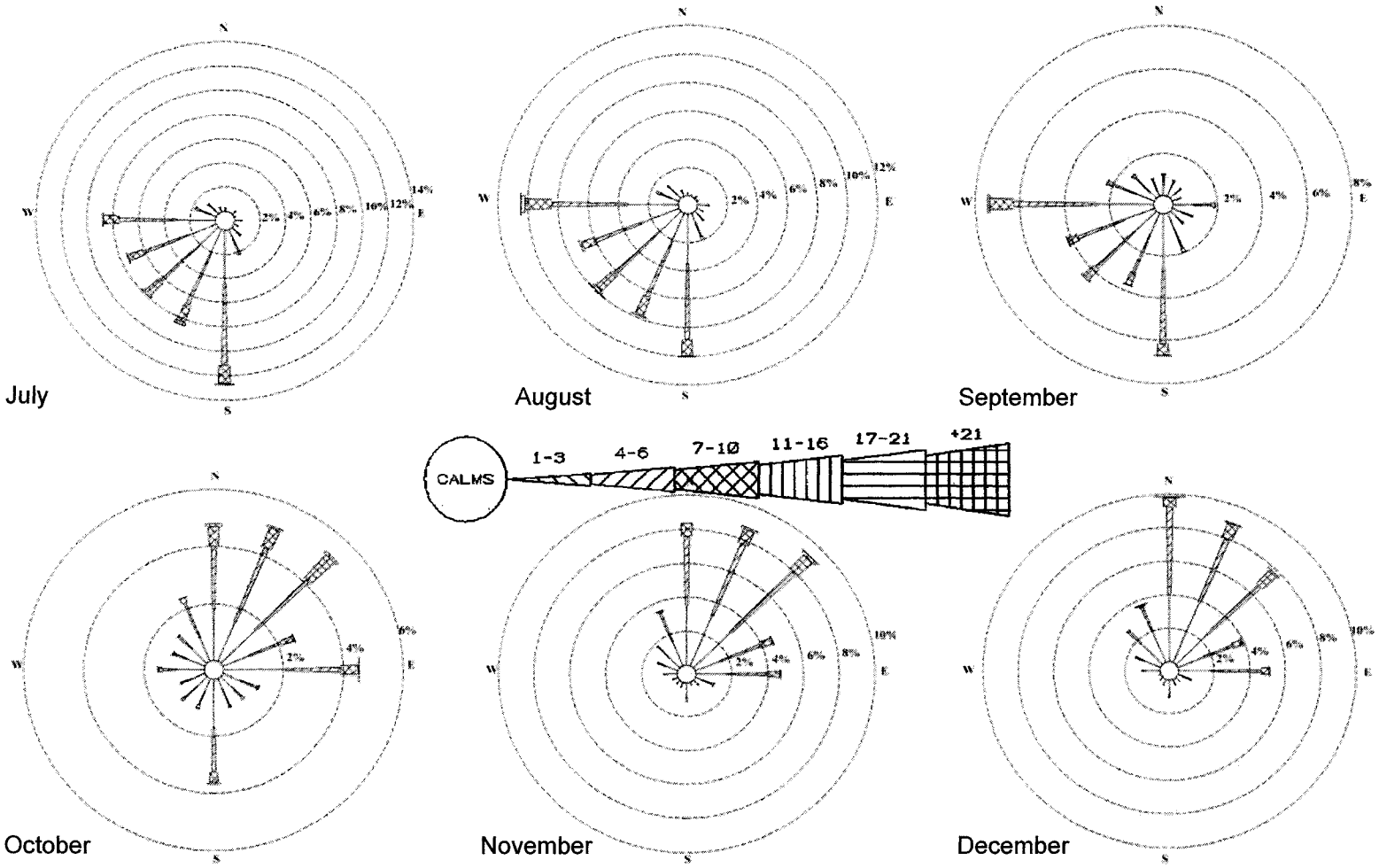


Figure A4. Wind roses based on 1991-2000 data at the Bangkok Metropolis Station
Note: Direction is from which the wind blows. Height of wind vane is 33 meters above ground. Wind Speed in knots
Calm wind frequency is 54%, 39%, 30%, 37%, 49%, 48%, 46%, 48%, 60%, 61%, 53%, and 52% for January, February, March, April, May, June, July, August, September, October, November, and December, respectively.

A.3 Bangkok Diurnal Variation Data

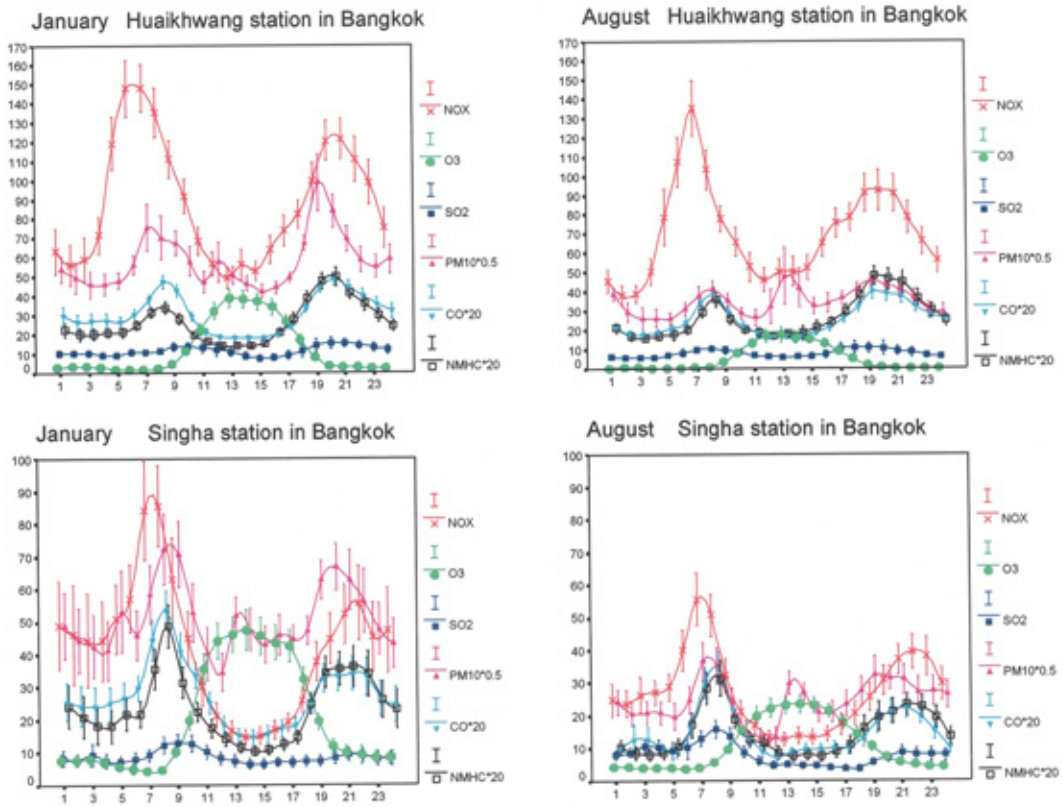


Figure A5 Average concentrations of pollutants at Huaikhwang and Singha stations in Bangkok city based on the data of 1996-2000. Vertical bars represent two standard errors of mean values. The values for NO_x, O_3, and SO_2 are in ppb. To obtain PM_{10} values ($\mu g/m^3$), the readings from graph should be multiplied by two. To obtain CO (ppm) and NMHC (ppm) the readings should be multiplied by 0.05.

Appendix B

Maptaphut Industrial Estate

B.1 Industries in Maptaphut Industrial Estate

Name of Factory and Their Product in Maptaphut Industrial Estate

1. Bangkok Sintitics Co, LTD. (BST); Products: Mixted C4 (MTBE, Butene-1 Butadiene)
2. Gran Siam Composit Co, LTD. (CSCS); Product: Plastics (PP Compound)
3. Kromton special Co, LTD.; Product: Rubber Chemical (Antidegradnts)
4. The Cogeneration Co, LTD. (COCO); Product: Current, Steam and Water for Industrial
5. TPC Pressresin Co, LTD. (TPC); Product: PVC Plastics Powder (Paste PVC Resin Type)
6. Tuntex (Thailand) Public Company. (TTC); Product: Tuntex Petrochemical (Thailand) Public company. (TPT); Product: Pure Terephthalic Acid (PTA)
7. Thai GCI Resitop CO, LTD. (TGCI); Product: Polyester Yarn
8. Thai Baroda Industry CO, LTD. (BTL)
9. Thai Cogeneration CO, LTD. (TCC); Product: Electric, Steam and Water
10. Thai Sinkong Industry CO, LTD. (TSIC); Product: Polyester Low IV and Botel Pet Rasin
11. Thai Plastics and Chemicals Public Company. (TPC); Product: PVC Plastics Powder, VCM, Sodium Bicarbonate
12. Thai Polypropylene CO, LTD. (TPP); Product: Polypropylene Bead
13. Thai Polypropylene (1994) CO, LTD. (TPP1994); Product: Polypropylene Bead
14. Thai Polyethylene CO, LTD. (TPE); Product: Polyethylene Bead (LLDPE/HDPE)
15. Thai Polyethylene (1993) CO, LTD. (TPE1993); Product: Polyethylene Bead (LDPE)
16. Thai wire Product CO, LTD. (TWP)
17. Thai Scandic Steel. (TSS)
18. Thai Industrial Gas Public Company. (TIG); Product: Nitrogen, Oxygen, Hydrogen and Argon
19. Thai Epoxy and Alline Product CO, LTD. (TEC); Product: Epoxy Resin
20. Thai MFC CO, LTD. (MFC)
21. Thai MMA CO, LTD. (TMMA)
22. Thai Olefins CO, LTD. (TOC)
23. Nova Steel CO, LTD. (NOVA)
24. Bangkok Cogeneration CO, LTD. (BCC); Product: Current and Steam
25. Bangkok Polyethylene CO, LTD. (BPE)
26. Bangkok Industrial Gas CO, LTD. (BIG1)
27. Bangkok Industrial Gas CO, LTD. (BIG2); Product: Nitrogen, Oxygen, Hydrogen
28. BST Erasetomert CO.LTD. (BST); Product: Butadiene Rubber, Stryrene Butadiene Rubber
29. Bayer Polymer CO, LTD. (BPC); Product: Plastics Bead (ABS/SAN/PC)
30. National Petrochemical CO, LTD. (NPC); Product:

31. National Fertilizer CO, LTD. (NFC)
32. Pornpat Chemical CO, LTD. (PORNPAT); Product: Silicon Dioxide
33. PPG Siam Silica CO, LTD. (PPG)
34. Peroxide Thai CO, LTD.; Product: Hydrogen Peroxide
35. United Silica (Siam) CO, LTD. (PPG); Product: Silicon Dioxide, Sodium Aluminium Silicate
36. Rayong Olefins CO, LTD. (ROC); Product: Ethylene, Propylene/ Mixed C4, Toluene, Benzene
37. Rayong Wire Industry CO, LTD. (RWI)
38. Laport (Thailand) CO, LTD.
39. Viny Thai CO, LTD. (VNT)
40. Sakchaisith CO, LTD.
41. Thai Vilas Steel CO, LTD.
42. Pacific Plastics (Thailand) CO, LTD. (PPTL); Product: Polyether Polyol and Formulated Polyal
43. Siam Polyethylene CO, LTD. (SPECL)
44. Siam Polystyrene CO, LTD. (SPCL); Product: Polystyrene Plastics Bead
45. Siam Synthetic Latex CO, LTD. (SSLC); Product: Styrene-Butadiene Latex
46. Siam Styrene Monomer CO, LTD. (SSMC); Product: Styrene Monomer and Toluene
47. Siam Contruction Steel CO, LTD. (SCSC)
48. Siam Yamoto Steel CO, LTD. (SYS)
49. Aromatics Thai CO, LTD. (ATC)
50. HC Starch CO, LTD. (HCST)
51. HMC Polymers CO, LTD. (HMC); Product: Polypropylene Plastics Bead
52. HMT Polystyrene CO, LTD. (HMT) Product: Polystyrene Plastics Bead
53. Liquid Air (Thailand) CO, LTD. (LA); Product: Industrial gas, Hydrogen, Steam and Carbon Dioxide
54. Star Petroleum Refining CO, LTD. (STAR); Product: Petroleum
55. Rayong Refinery CO, LTD. (RRC); Product: Petroleum
56. G and Environmental Natural CO, LTD. (GENCO)
57. Kartoon Natee Saemcorp CO, LTD. (KNS); Product: Warehousing, Logistics, Storage, Silostorage and Handing
58. Thai Tank Terminal CO, LTD. (TTT): Port and Chemical Cargo
59. Rayong Port CO, LTD. (RBT)
60. MTP Cogeneration CO, LTD. (MTP)
61. Map Ta Put Tank Terminal CO, LTD. (MTT)
62. Thai Prosperity CO, LTD. (TPT-P)
63. Eastern Fluid Transport CO, LTD. (EFT)

B.2 Types of Industry and related pollutants

Grain processing/storage
Main Pollutants: dust and combustion pollutants
The plants in this category are mainly silos and rice mills. The main air pollution sources are from the transport of grain, the use of rice husks and shells to produce energy, which can cause black smoke. There are more than 200 rice mills in Rayong Area.

Incinerators
Main Pollutants: dust and combustion pollutants
Incinerators are found in the industrial estates. Saraburi, Chon Buri and Rayong all have industrial estates with incinerators. Most of them have 500 kg/hr capacity, but each estate has several of them. All of them are multi chamber type, using diesel oil and have air pollution test records which were found to be a very clean operation. The hospitals in each province have at least one incinerator, but were found to be rarely used.

Canning Industry
Main Pollutants: combustion pollutants, odors
The canning industry is for local agricultural products and is seasonal in nature. The boiling of agricultural products was found to consume small amounts of fuel and the odor is mainly from wastewater.

Edible oil extraction
Main Pollutants: combustion pollutants, odors
Air pollution associated with this industry is mainly odor from wastewater. Fuel oil is used to generate the steam. There are very few factories in the Rayong area.

Food additives/Bakeries
Main Pollutants: combustion pollutants, odors

Fishmeal and bonemeal
Main Pollutants: dust, odor, combustion pollutants
This type of factory has not generated many complaints regarding odors, because the provinces, such as Chonburi and Rayong, have been successful in relocating these plants to appropriate areas. The quality of operation, such as using fresh fish, has been successful so that the problem of odors has been subsiding.

Textiles
Main Pollutants: combustion pollutants
Textile plants are usually very large consumers of fuel oil, but in the Rayong area most of them are local, such as silk spinning and are small operations.

Pulp and paper
Main Pollutants: combustion pollutants, odors
The problem is mainly characteristic odors.

Agroindustry
Main Pollutants: dust, combustion pollutants, odors
Tapioca pelletization factories are the most numerous of all sources. The common cause of air pollution is odor. Dust problems have subsided due to better control. The main problem is now the tapioca flour industry with wastewater odors. The whiskey industry also has a problem of odor from its wastewater treatment plants. The tobacco curing industry has been found to decline drastically in the North, with the closure of more than 80% of the curing houses, which used to cause air pollution due to the use of lignite.

Plastics and Chemicals
Main Pollutants: combustion products, odors
This type of industry is found exclusively in Rayong.

Oil refineries
Main Pollutants: dust, combustion products, odors
The major emissions are mainly sulfur dioxide and oxides of nitrogen, as well as hydrocarbons.

Ceramics
Main Pollutants: dust, combustion products
Ceramic industry in Thailand uses mainly LPG as fuel. Some of them still use firewood for low quality ceramics.

Glass
Main Pollutants: dust, combustion products
Glass factories use fuel oil which is the cause of emissions.

Mineral
Main Pollutant: dust
Rock crushing plants cause severe dust problems.

Metal
Main Pollutants: dust, smelting products, combustion pollutants
All of them have baghouses as a pollution control.

Machinery production
Main Pollutants: odors, combustion pollutants
This includes car and bus manufactures. The problem is volatile material used in paints.

Power generation

Main Pollutants: combustion pollutants, dust

Gas Separation
Main Pollutants: odors, combustion pollutants

B.3 Photographs of Maptaphut Industrial Estates

Figure B.1 East-(North-East) View from Central Tower (Zoom).

Figure B.2 East-(North-East) View from Central Tower.

Figure B.3 North-East View from Central Tower.

Figure B.4 North-West View from Central Tower.

Figure B.5 North-(North-East) View from Central Tower.

Figure B.6 North-(North-East) View from Central Tower.

Figure B.7 North-(North-East) View from Central Tower (Zoom).

Figure B.8 East View from Central Tower.

Figure B.9 South-East View from Central Tower (Zoom), Maptaphut Industrial Port behind.

Figure B.10 East-(South-East) View from Central Tower.

Figure B.11 Maptaphut Industrial Port (Zoom).

Figure B.12 South-(South-East) View from Central Tower.

Figure B.13 South-West View from Central Tower.

Figure B.14 West View from Central Tower.

Figure B.15 North View from Central Tower.

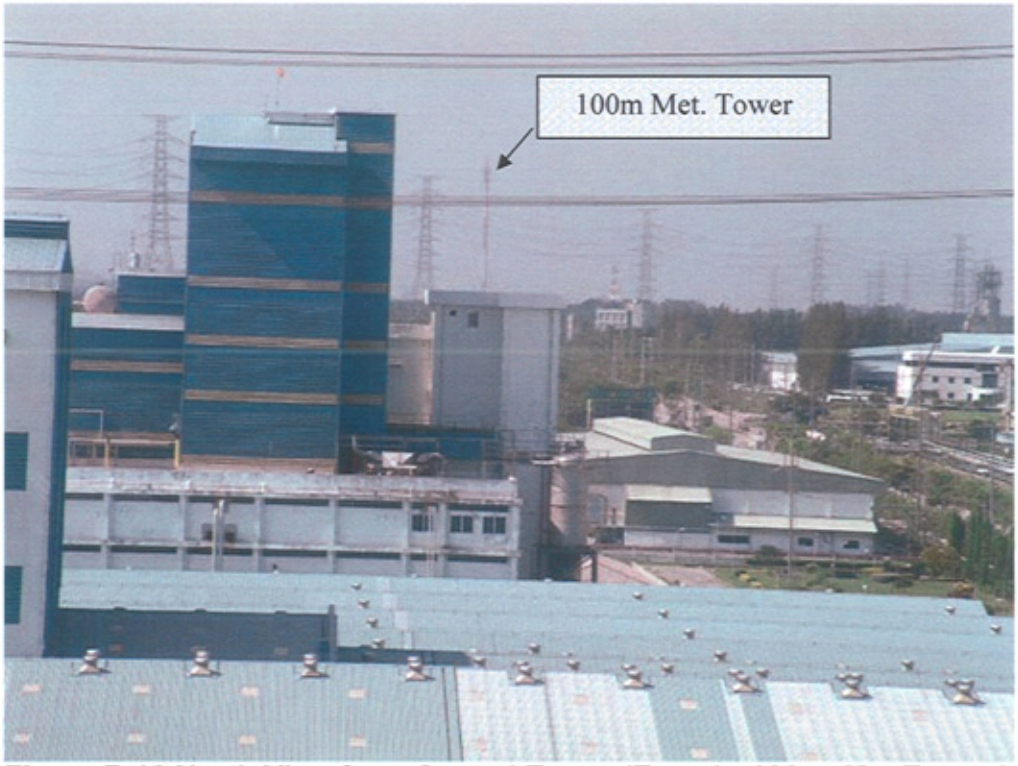


Figure B.16 North View from Central Tower (Zoom), 100 m Met Tower behind.

B.4 Arial Map Photo of Maptaphut Industrial Estate

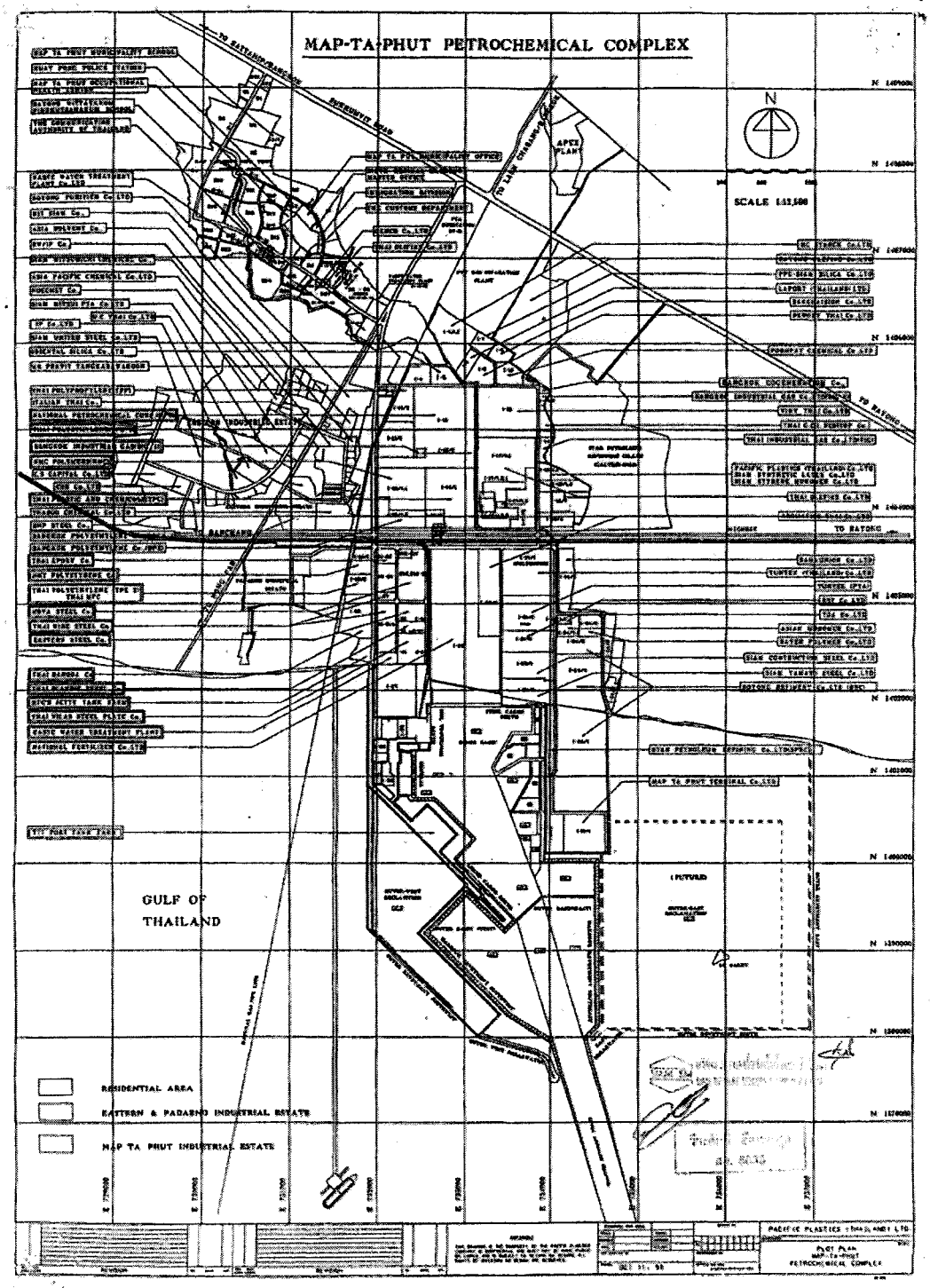


Figure B.17 Maptaphut Industrial Estates.

Figure B.18 Left Portion of Maptaphut Industrial Estates.
(Source: Google Earth)

Figure B.19 Middle Portion of Maptaphut Industrial Estates.
(Source: Google Earth)

Figure B.20 Right Portion of Maptaphut Industrial Estates. (Source: Google Earth)

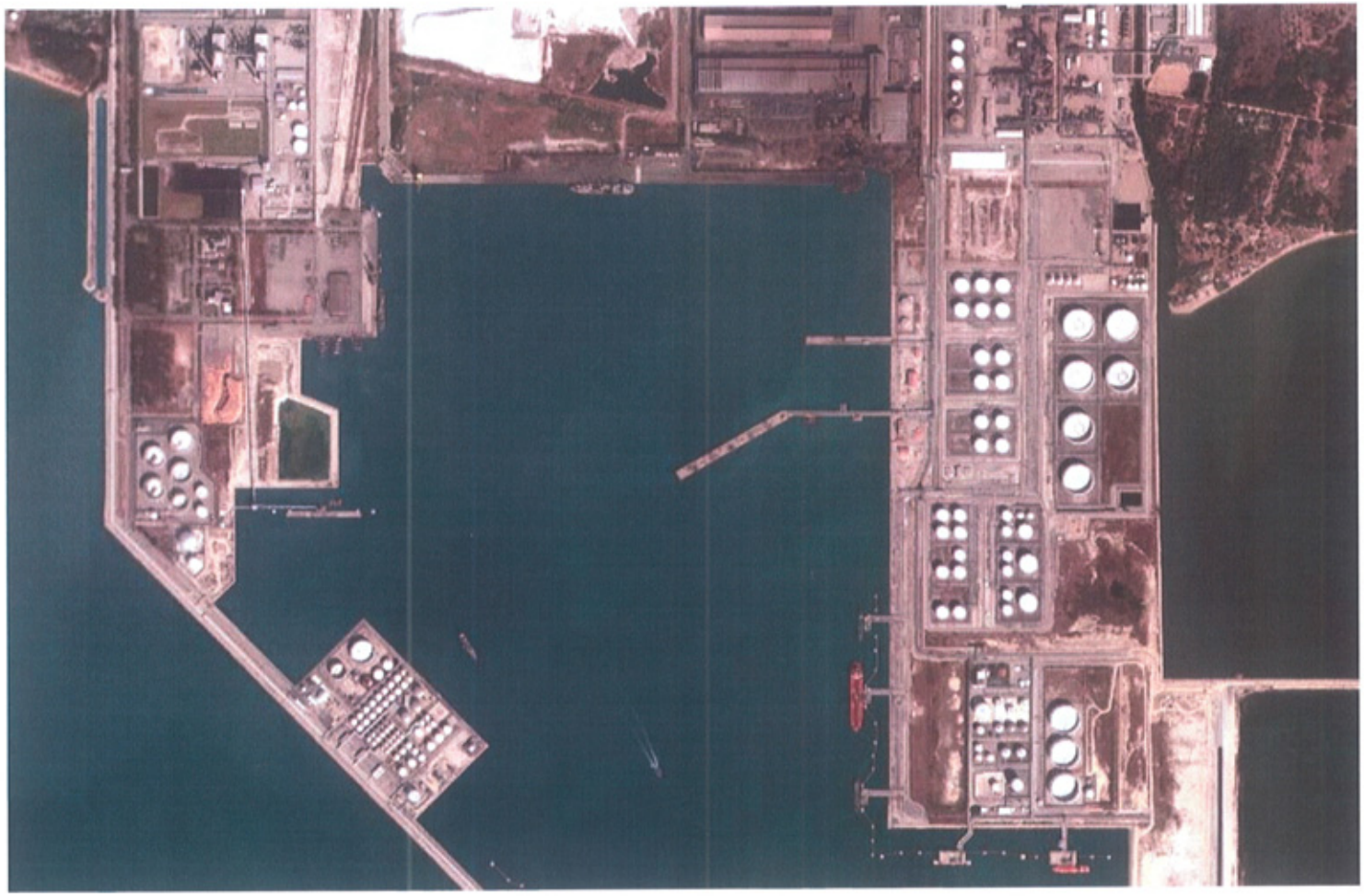


Figure B.21 Lower Portion of Maptaphut Industrial Estates.
(Source: Google Earth)

Appendix C

Format of Raw Data Stored on PCD Data Base Computer
(Ambient and Meteorological Data)

Listing C.1 Format of Raw data store on PCD data base computer (Ambient Data)

```
##################################################################
###########
#
# Graph type: time series
#
# Database type: 1 Hour
#
# 1/1/98 00 - 1/1/98 00
#
# Selected time series:
# Rayong 31T, CO ppm, 003, Value
# Rayong 31T, O3 ppb, 003, Value
# Rayong 31T, SO2 ppb, 003, Value
# Rayong 31T, PM10, 003, Value
#
#Cut condition:
#
#
# Weekday : All
# Day type: All
# Period in year (mmdd): 0101 - 1231
# Period within day(hh): 01 - 24
# Moving average: 1(1)
#
#
# Plot var. :
# 1. x1:
# 2. x2:
# 3. x3:
# 4. x4:
#
#IGNORE: This is a way to ignore all data
ZAP+ 0 # Just to comply to the standard
EOH
#DATA
991119, 0100,      0.4,      30,       7,      22,
991119, 0200,        ,       ,         ,      30,
991119, 0300,      0.4,      33,       7,      32,
991119, 0400,      0.4,      33,       6,      30,
991119, 0500,      0.4,      35,       4,      37,
991119, 0600,      0.4,      34,       5,      35,
991119, 0700,      0.5,      34,       2,      40,
991119, 0800,      0.5,      37,       2,      57,
991119, 0900,      0.5,      40,       1,      52,
991119, 1000,   0.599999,      36,       2,      39,
991119, 1100,      0.5,      30,       1,      46,
991119, 1200,      0.5,      41,       1,      45,
991119, 1300,      0.5,      42,       1,      46,
991119, 1400,      0.5,      43,       2,      49,
991119, 1500,      0.5,      36,       1,      39,
991119, 1600,   0.599999,      32,       2,      25,
991119, 1700,   0.599999,      27,       3,      27,
991119, 1800,   0.599999,       9,       2,      40,
991119, 1900,      0.5,      16,       6,      62,
991119, 2000,      0.4,       7,       8,      87,
```

```
991119, 2100,      0.4,      13,      18,      120,
991119, 2200,      0.4,      20,      10,      131,
991119, 2300,      0.4,      21,      23,      117,
991119, 2400,      0.4,      29,       8,       88,
```

Listing C.2 Format of Raw data store on PCD data base computer (Meteorological data)

```
##########################################################################
###########
#
# Graph type: time series
#
# Database type: 1 Hour
#
# 1/1/98 00 - 1/1/98 00
#
# Selected time series:
# Rayong 31T, Net rad, 003, Value
# Rayong 31T, Glob rad, 003, Value
# Rayong 31T, Pressure, 003, Value
# Rayong 31T, Rain, 003, Value
#
#Cut condition:
#
#
# Weekday : All
# Day type: All
# Period in year (mmdd): 0101 - 1231
# Period within day(hh): 01 - 24
# Moving average: 1(1)
#
#
# Plot var. :
# 1. x1:
# 2. x2:
# 3. x3:
# 4. x4:
#
#IGNORE: This is a way to ignore all data
ZAP+ 0 # Just to comply to the standard
EOH
#DATA
980103, 0100,      -17,       2,      760,       0,
980103, 0200,      -17,       2,      760,       0,
980103, 0300,      -15,       2,      759,       0,
980103, 0400,      -13,       2,      759,       0,
980103, 0500,      -12,       2,      759,       0,
980103, 0600,      -11,       2,      759,       0,
980103, 0700,       -9,       8,      760,       0,
980103, 0800,       14,      56,      761,       0,
980103, 0900,       70,     246,      761,       0,
980103, 1000,      211,     555,      762,       0,
980103, 1100,      263,     706,      762,       0,
980103, 1200,      288,     679,      762,       0,
980103, 1300,      263,     892,      761,       0,
980103, 1400,      368,     798,      760,       0,
980103, 1500,      322,     695,      760,       0,
```

```
980103, 1600,   212,   485,   759,   0,
980103, 1700,    80,   263,   760,   0,
980103, 1800,   -13,    54,   760,   0,
980103, 1900,   -29,     3,   760,   0,
980103, 2000,   -26,     2,   760,   0,
980103, 2100,   -26,     2,   761,   0,
980103, 2200,   -24,     2,   761,   0,
980103, 2300,   -20,     2,   761,   0,
980103, 2400,   -24,     2,   761,   0,
980104, 0100,   -20,     2,   761,   0,
980104, 0200,   -17,     2,   760,   0,
980104, 0300,   -15,     2,   760,   0,
980104, 0400,   -13,     2,   760,   0,
980104, 0500,   -13,     2,   760,   0,
980104, 0600,   -13,     2,   760,   0,
980104, 0700,    -9,     7,   760,   0,
980104, 0800,    15,    63,   761,   0,
980104, 0900,    73,   247,   762,   0,
980104, 1000,   194,   502,   762,   0,
980104, 1100,   232,   619,   762,   0,
980104, 1200,   361,   828,   762,   0,
980104, 1300,   264,   887,   761,   0,
980104, 1400,   381,   825,   760,   0,
980104, 1500,   310,   667,   760,   0,
980104, 1600,   186,   428,   760,   0,
980104, 1700,    83,   252,   760,   0,
980104, 1800,    -7,    55,   760,   0,
980104, 1900,   -25,     3,   760,   0,
980104, 2000,   -25,     2,   760,   0,
980104, 2100,   -25,     2,   760,   0,
980104, 2200,   -23,     2,   760,   0,
980104, 2300,   -20,     2,   761,   0,
980104, 2400,   -15,     2,   761,   0,
980105, 0100,   -19,     2,   761,   0,
980105, 0200,   -17,     2,   760,   0,
980105, 0300,   -16,     2,   760,   0,
980105, 0400,   -15,     2,   759,   0,
980105, 0500,   -12,     2,   759,   0,
980105, 0600,   -12,     2,   760,   0,
980105, 0700,    -8,     7,   760,   0,
980105, 0800,    15,    59,   761,   0,
980105, 0900,    67,   229,   761,   0,
980105, 1000,   199,   517,   762,   0,
980105, 1100,   226,   578,   762,   0,
980105, 1200,   272,   635,   761,   0,
980105, 1300,   183,   501,   761,   0,
980105, 1400,   108,   260,   760,   0,
```

Appendix D

Accuracy of Daily Maximum Ozone and PM_{10} by Statistical Model Calculation

Table D.1 Daily maximum monitoring data and calculate data of Ozone year 1998-2004 during dry season (not include wet season, May to September) at 29, 30, 31T station with different between both data and error percentage.

	Observe	Calculate	Diff	%Error
Sum	56,762.00	56,159.63	-602.37	31,136.32
N	1,478.00	1,478.00	1,478.00	1,478.00
Avg.	38.40	38.00	-0.41	21.07

Note:

$$\text{Diff} = \text{Calculate} - \text{Observe}$$

Equation D.1

$$\%\text{Error} = \frac{\text{Diff} * 100}{\text{Observe}}$$

Equation D.2

Observe is the daily maximum Ozone monitoring data at 29, 30, 31T station year 1998 – 2004 during dry season (not include wet season, May to September)

Calculate is the output of the multiple linear regression equation that derived from 5-year average consideration (1998-2002)

$$\mathbf{O_3 = 74.1 - 0.7\ NO + 0.5\ NO_2 + 3.4\ CH_4 + 10.7\ NMHC - 0.6\ RH - 5.6\ WS10}$$

Equation D.3

Observed Ozone = daily maximum hourly Ozone during 12.00 – 16.00 pm, ppb
NO, NO_2 = NO, NO_2 at 7.00 am, ppb
CH_4, NMHC = CH_4, NMHC at 7.00 am, ppm
RH = Relative Humidity at 7.00 am, %
WS10 = wind speed at 10 m at 7.00 am, m/s

Source of data are from the ambient air quality monitoring station, except meteorological data of 30T station are from meteorological station 100 meters tall.

NMHC and CH_4 data measured at 30T station are not available since 2003 due to no HC analyzer.

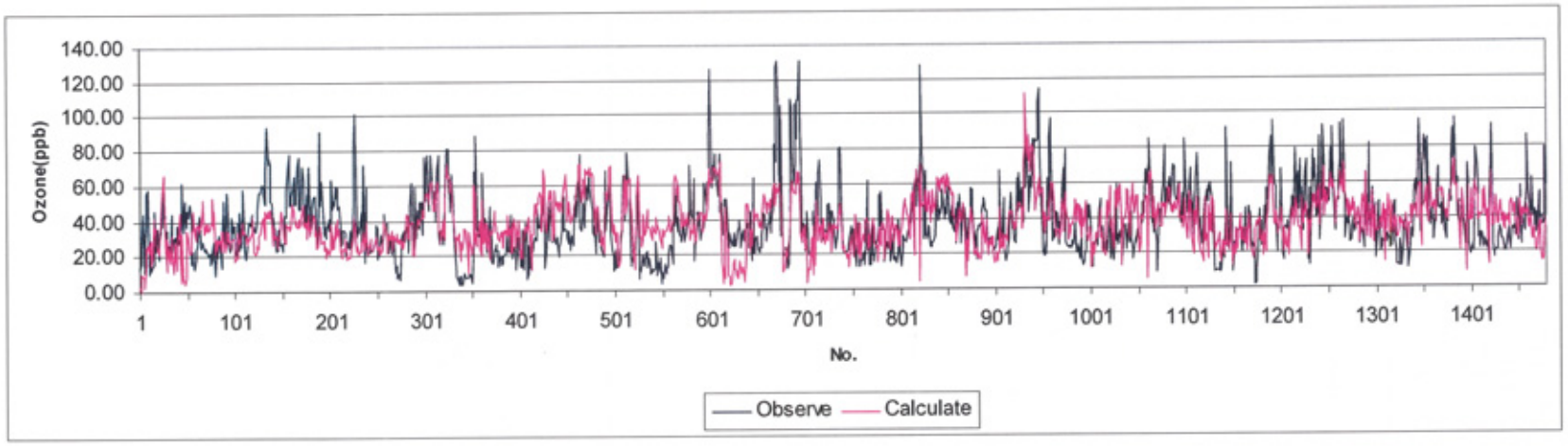


Figure D.1 Daily maximum observe and calculate Ozone data during dry season year 1998 to 2004 at 29, 30, 31T station at Rayong province, Thailand

The figure of comparison between daily maximum Ozone observed and calculated data presents the same direction trend of both data. However, some calculated data are not fit correctly with the monitoring data because the predicted data is depend on many parameters in both meteorological parameters and air quality parameters, which change all the time. Therefore, the equation can predict trend of the daily maximum Ozone at a satisfied level with 21 % error.

Table D.2 Daily maximum monitoring data and calculate data of PM10 year 1998-2004 during dry season (not include wet season, May to September) at 29, 30, 31T station with different between both data and error percentage

	Observe	Calculate	Diff	%Error
Sum	112,627.00	108,355.32	-4,271.67	67,007.28
N	1,754.00	1,754.00	1,754.00	1,754.00
AVG	64.21	61.78	-2.44	38.20

Note:

$$\text{Diff} = \text{Calculate} - \text{Observe}$$

Equation D.4

$$\%\text{Error} = \frac{\text{Diff} * 100}{\text{Observe}}$$

Equation D.5

Observe is the PM-10 monitoring data at 29, 30, 31T station year 1998 – 2004 during dry season (not include wet season, May to September)

Calculate is the output of the multiple linear regression equation that derived from 5-year average consideration (1998-2002)

$$\mathbf{PM_{10} = 305.8 + 26.6\ CO - 0.7\ RH - 5.3\ T + 27.1\ WS10 - 52.1\ WS50 + 24.9\ WS100}$$

Equation D.6

Observed PM_{10} = maximum 1-h PM_{10} during 6.00 – 10.00 am, $\mu g/m^3$

CO = CO at the same time as maximum PM_{10} occurred, ppm

RH = Relative Humidity at the same time as maximum PM_{10} occurred, %

T = temperature at the same time as maximum PM_{10} occurred, degree C

WS10, 50, 100 = wind speed at 10, 50, 100 m at the same time as maximum PM_{10} occurred, m/s

Source of data are from the ambient air quality monitoring station (29, 30, 31T), except meteorological data of 30T station and WS10, WS50, WS100 are from meteorological station 100 meters tall.

The equation results are not fit with the monitoring data correctly due to small number of PM10 monitoring data and also including CO, RH, and Temperature data during 1998 – 2002 to input in multiple linear regression model. For more accuracy, next research should determine the prediction equation by large number of monitoring data.

The comparison between observed data and predicted data by the equation is presented in the following figure.

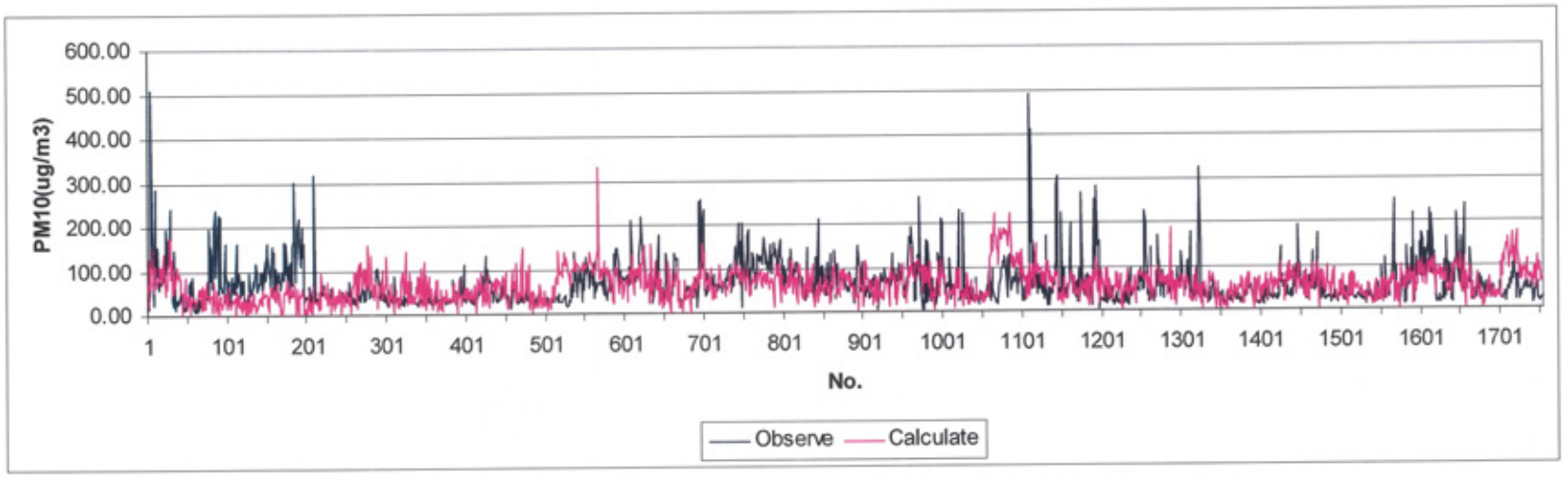


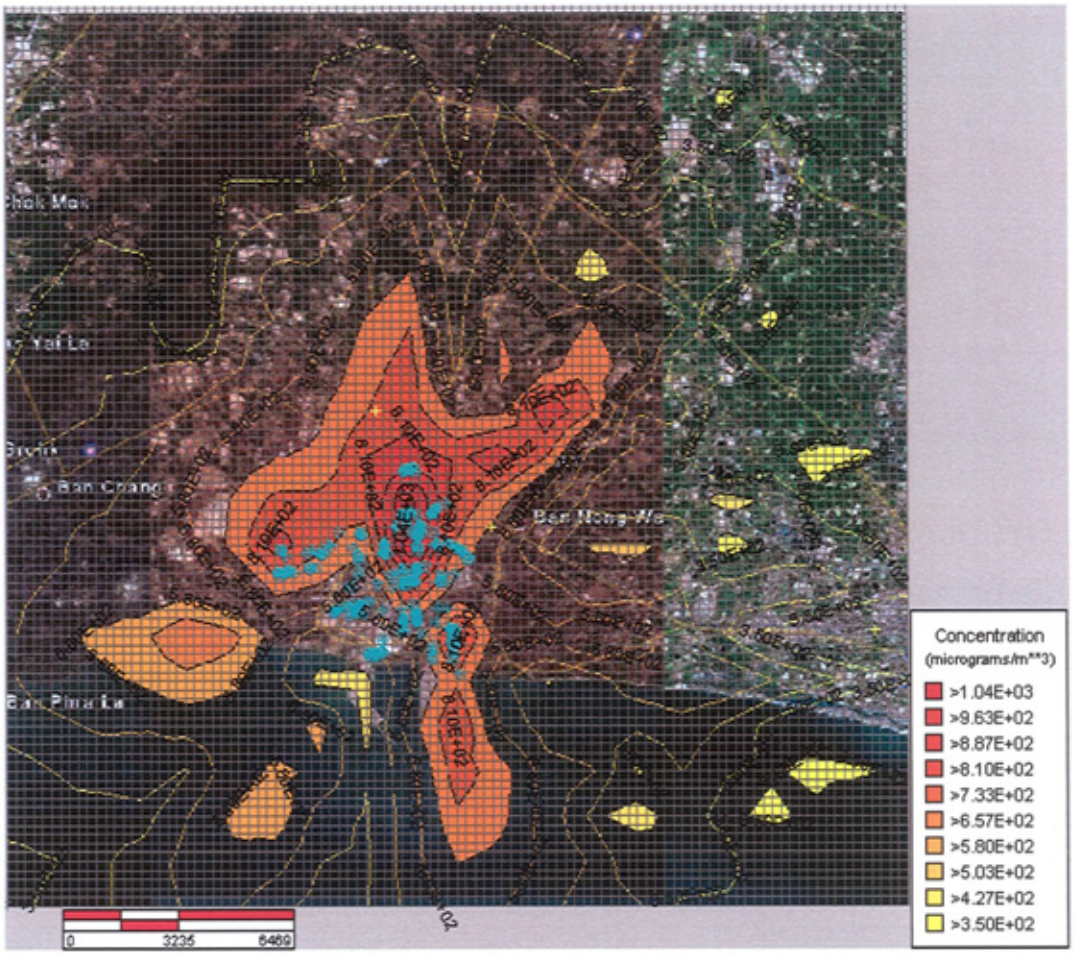
Figure D.2 Daily maximum observed and calculated PM10 data during dry season year 1998 to 2004 at 29, 30, 31T station at Rayong province, Thailand

According to the figure of comparison between observed and calculated data, trend of both data are in the same direction. However, some calculate data are higher or lower because the predicted data is depend on many parameters in both meteorological parameters and air quality parameters, which vary all the time. Therefore, the equation can predict trend of the daily maximum PM_{10} at a satisfied level when lacking of PM_{10} measurement data with 38 % error.

Appendix E

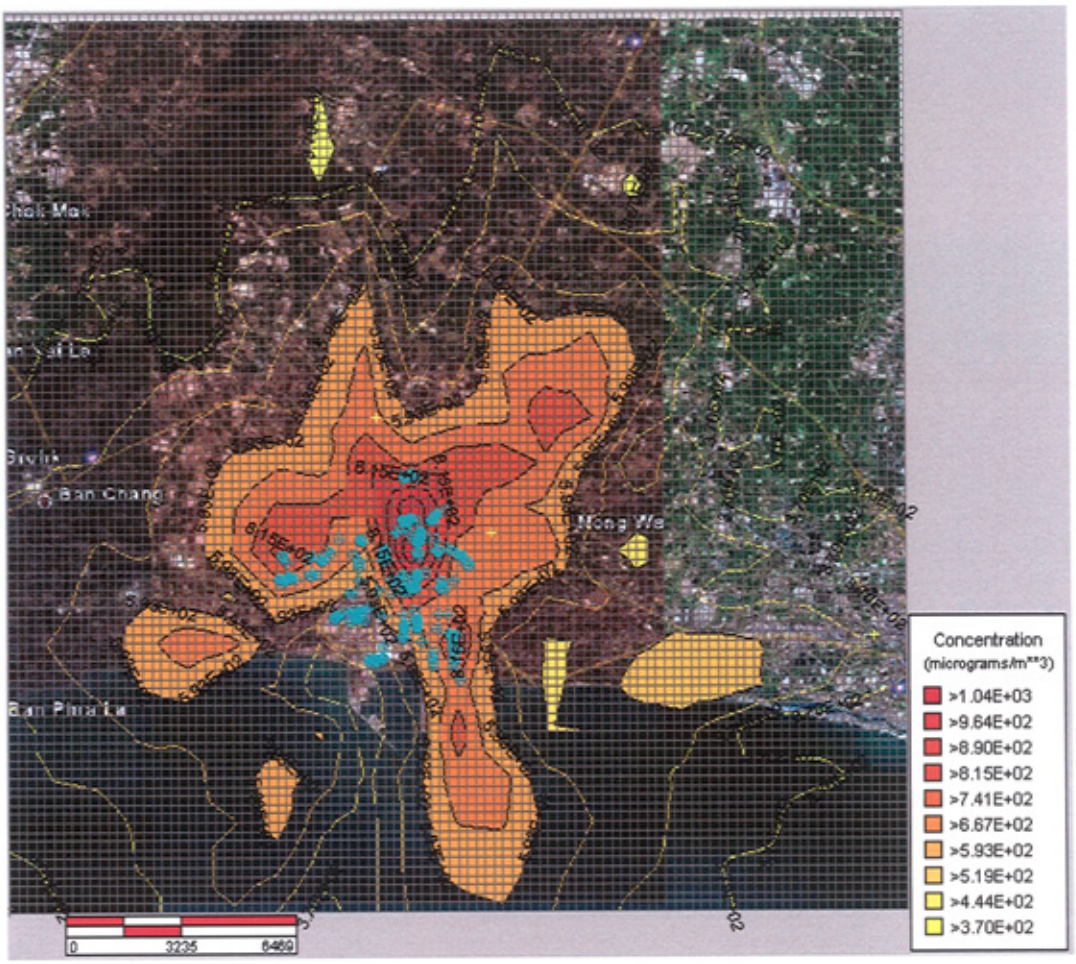
Seasonal ISC Contour Plots

E.2 Dry and Wet Season Contour Results of SO_2 dispersion by ISC model



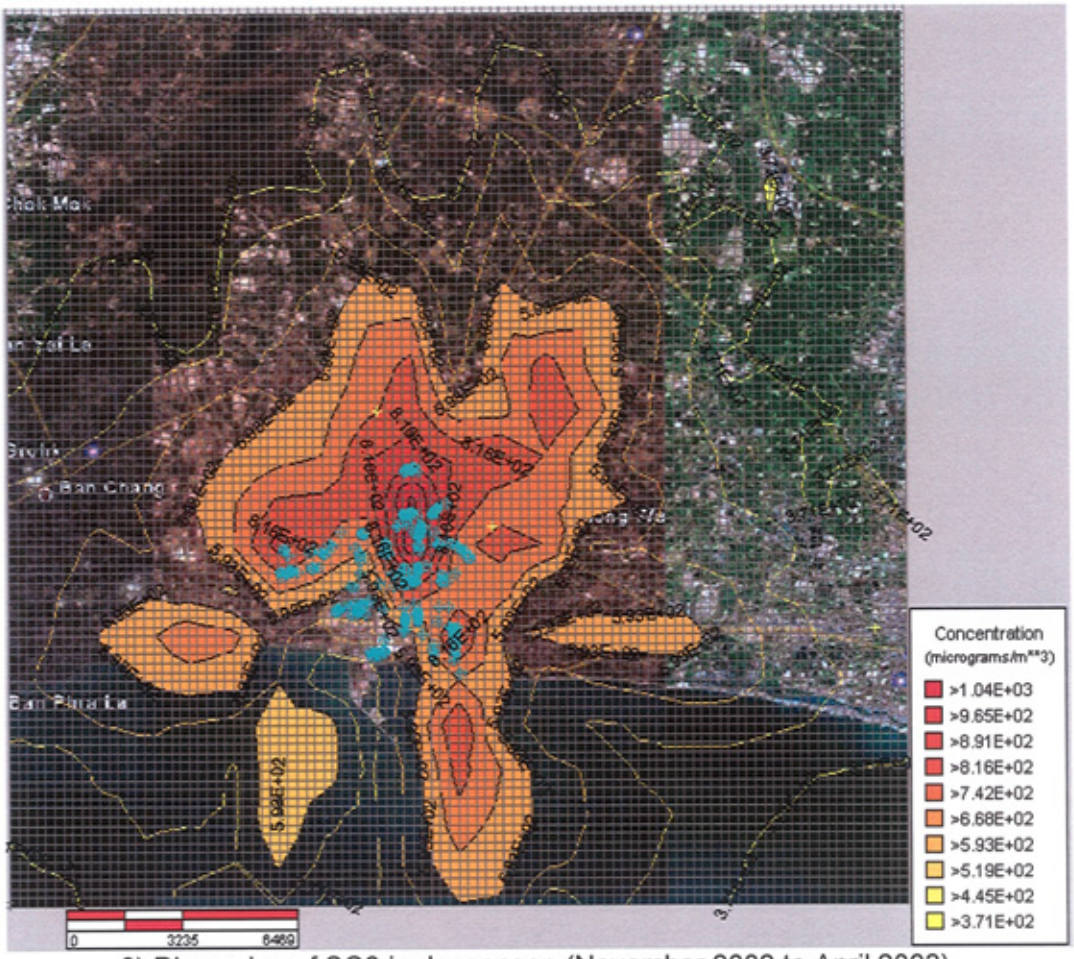
1) Dispersion of SO2 in dry season (January to April), 2002

Figure E.1 ISC model Result of wet and dry season SO2 concentration dispersion from the Maptaphut Industrial Estate.



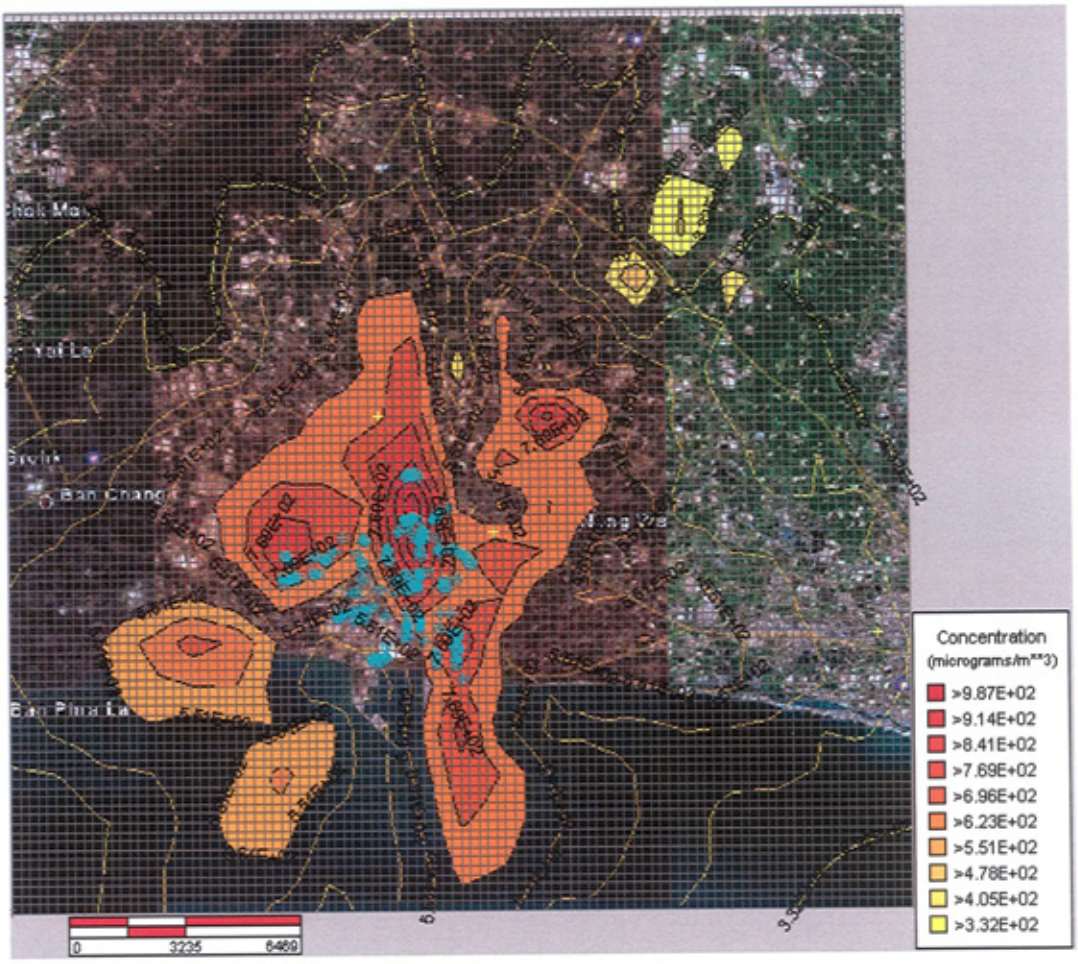
2) Dispersion of SO2 in wet season (May to October), 2002

Figure E.1 ISC model Result of wet and dry season SO2 concentration dispersion from the Maptaphut Industrial Estate (continued).



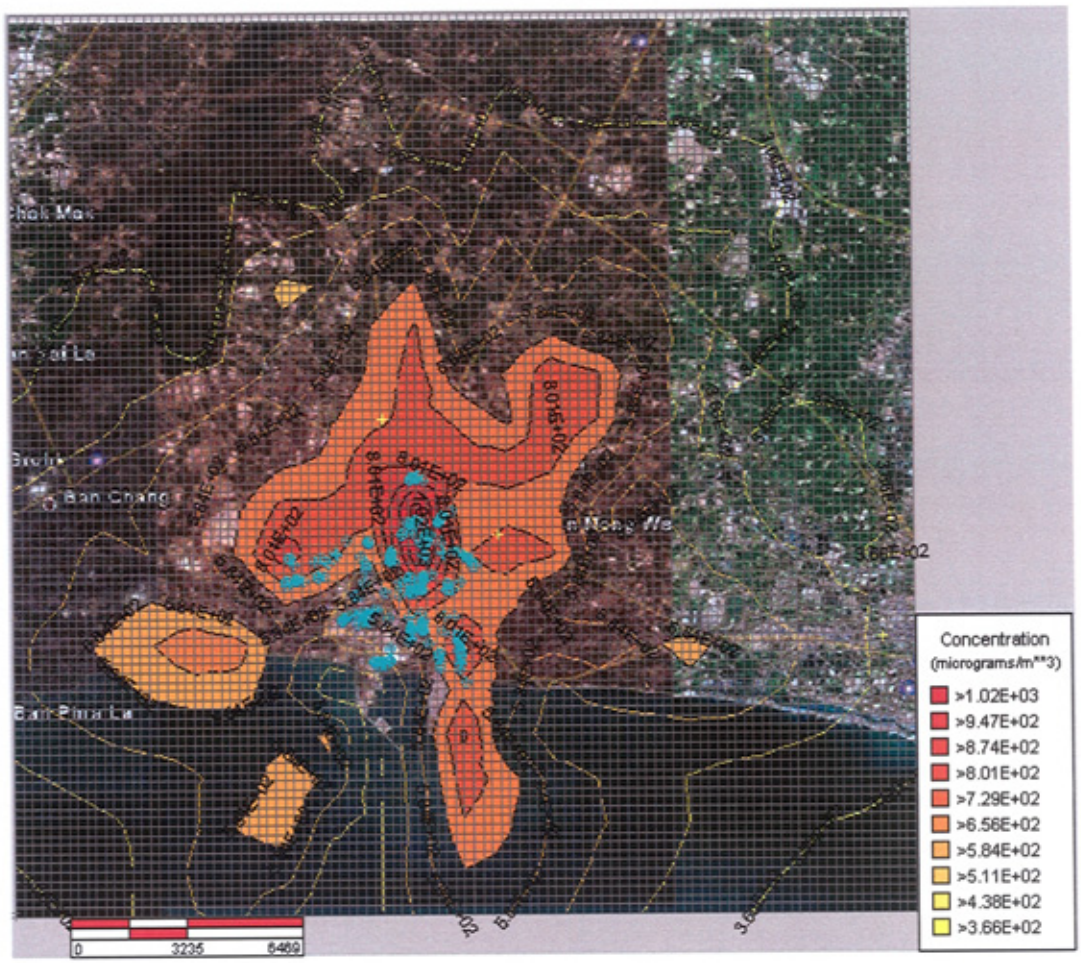
3) Dispersion of SO2 in dry season (November 2002 to April 2003).

Figure E.1 ISC model Result of wet and dry season SO2 concentration dispersion from the Maptaphut Industrial Estate (continued).



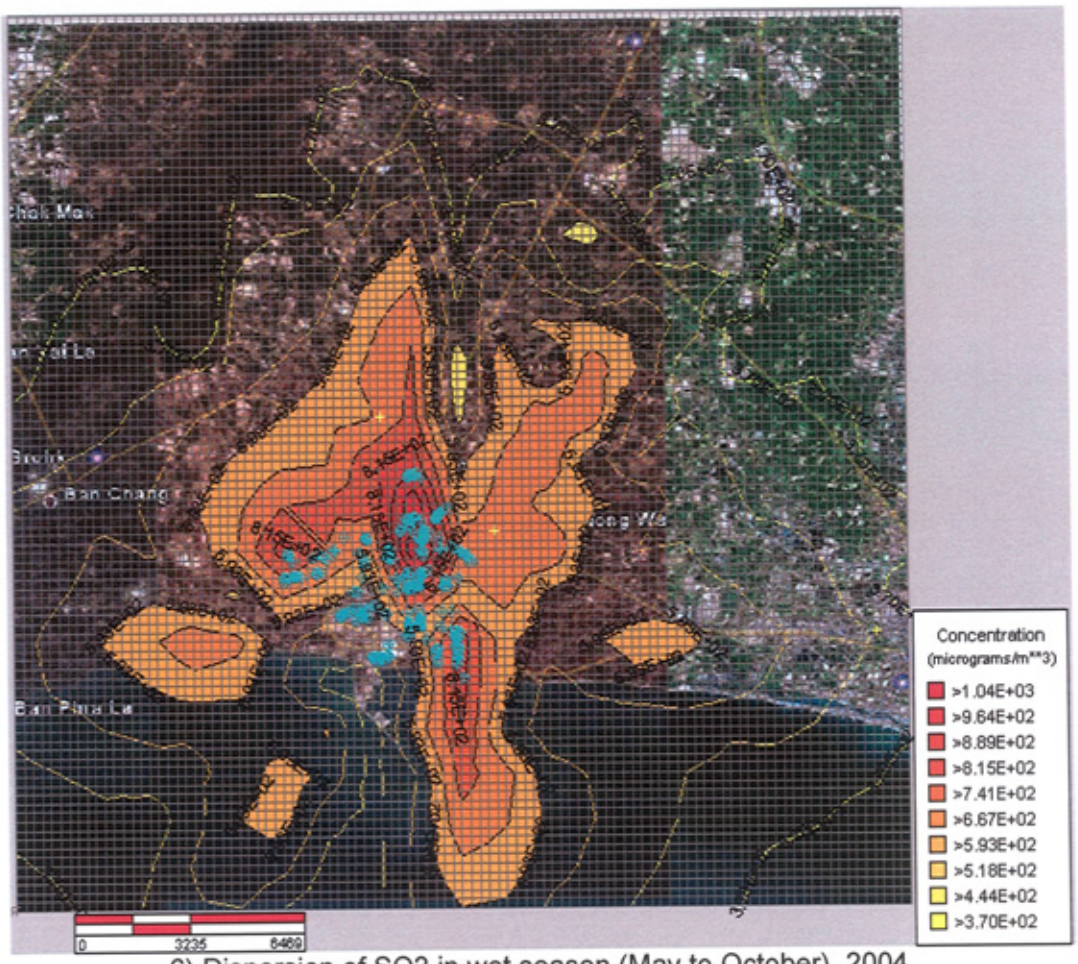
4) Dispersion of SO2 in wet season (May to October), 2003

Figure E.1 ISC model Result of wet and dry season SO2 concentration dispersion from the Maptaphut Industrial Estate (continued).



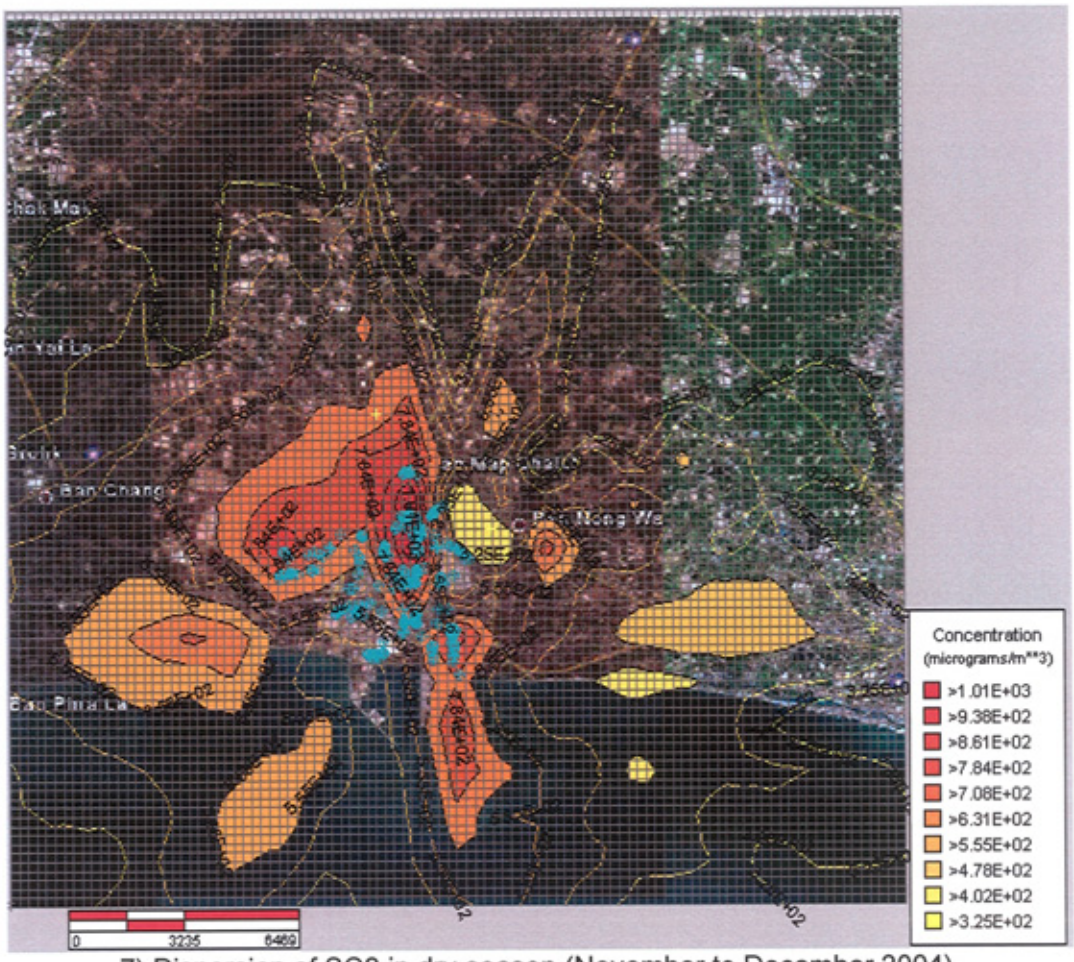
5) Dispersion of SO2 in dry season (November 2003 to April 2004).

Figure E.1 ISC model Result of wet and dry season SO2 concentration dispersion from the Maptaphut Industrial Estate (continued).



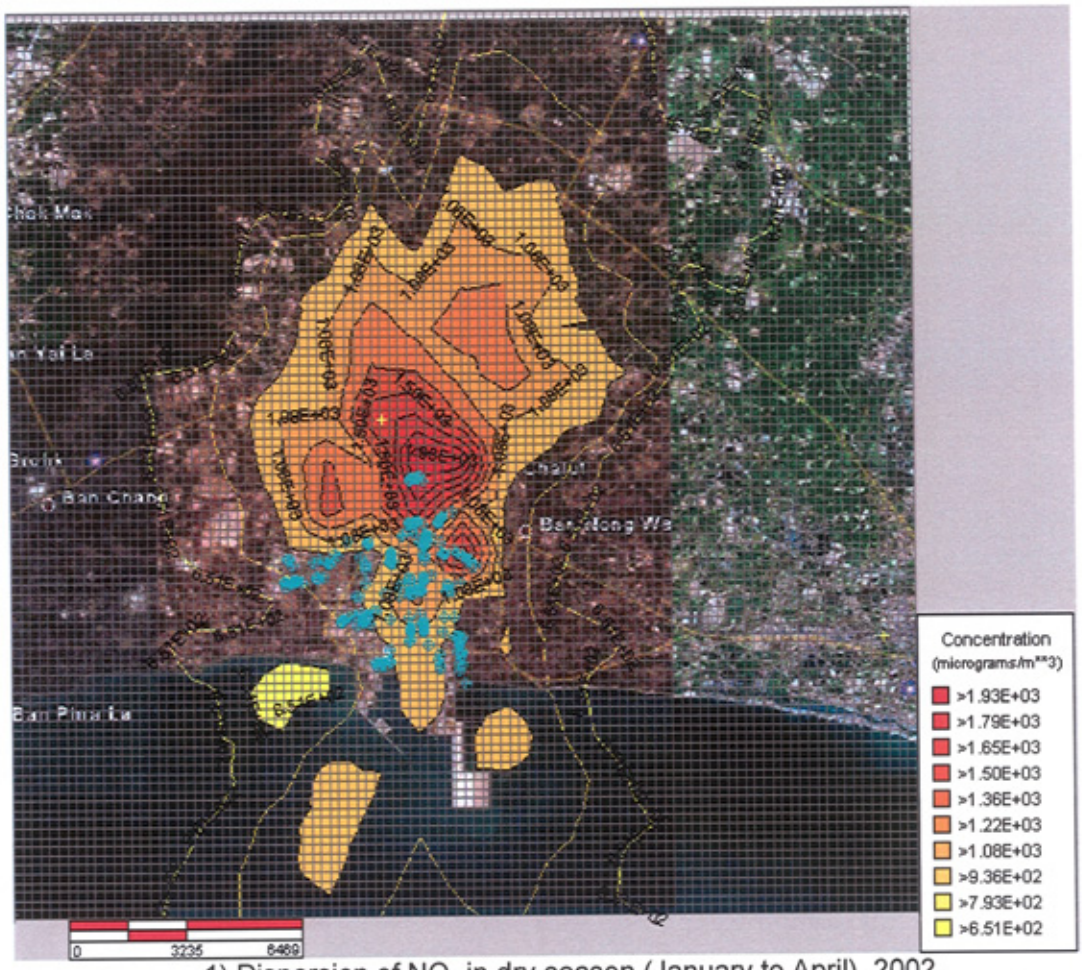
6) Dispersion of SO2 in wet season (May to October), 2004.

Figure E.1 ISC model Result of wet and dry season SO2 concentration dispersion from the Maptaphut Industrial Estate (continued).



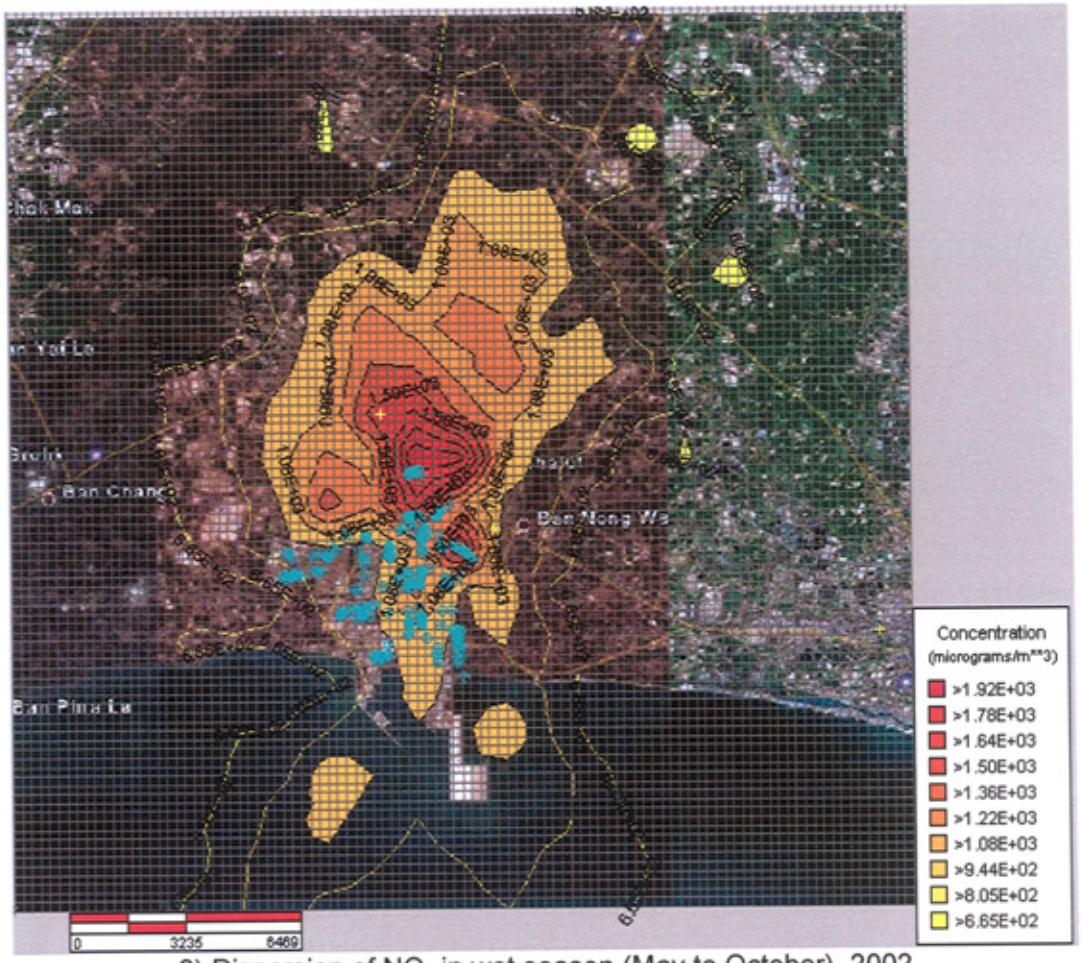
7) Dispersion of SO2 in dry season (November to December 2004).

Figure E.1 ISC model Result of wet and dry season SO2 concentration dispersion from the Maptaphut Industrial Estate (continued).



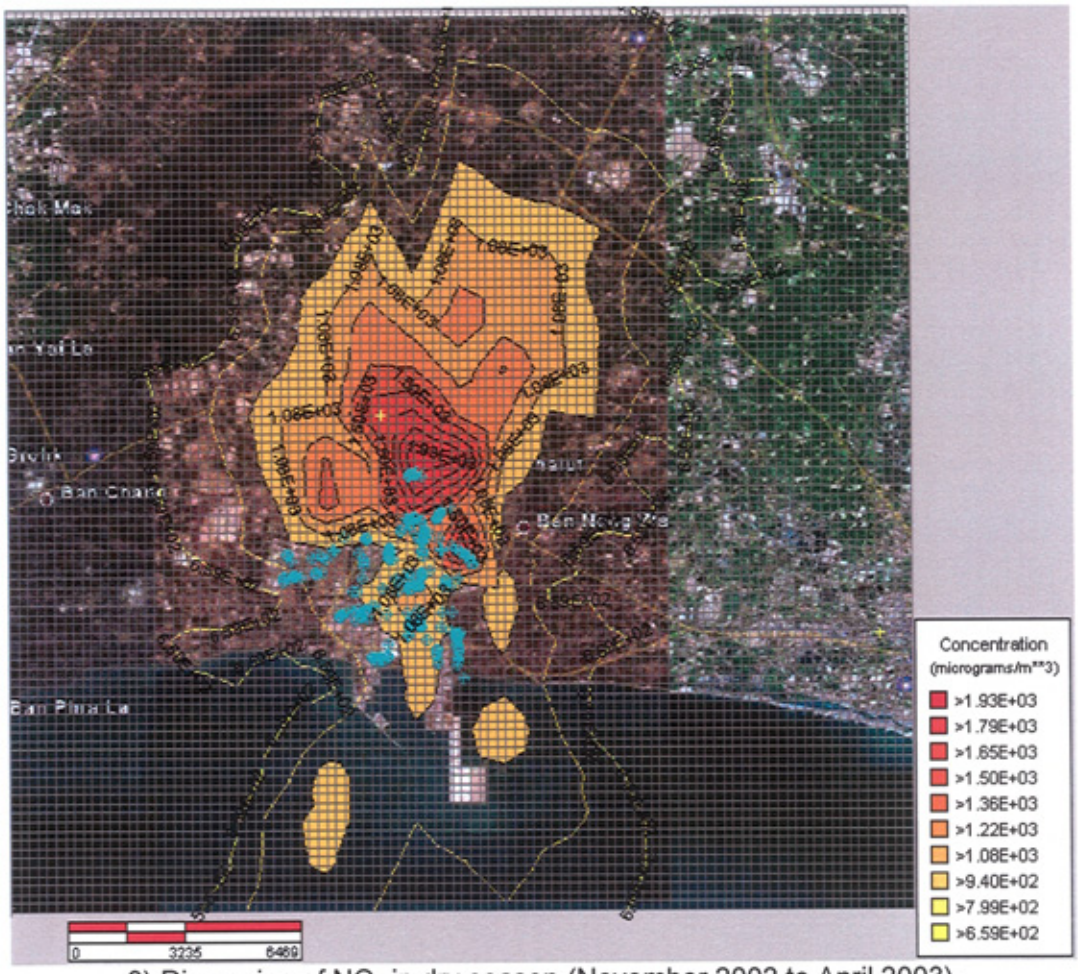
1) Dispersion of NO_x in dry season (January to April), 2002

Figure E.2 ISC model Result of wet and dry season NO_x concentration dispersion from the Maptaphut Industrial Estate.



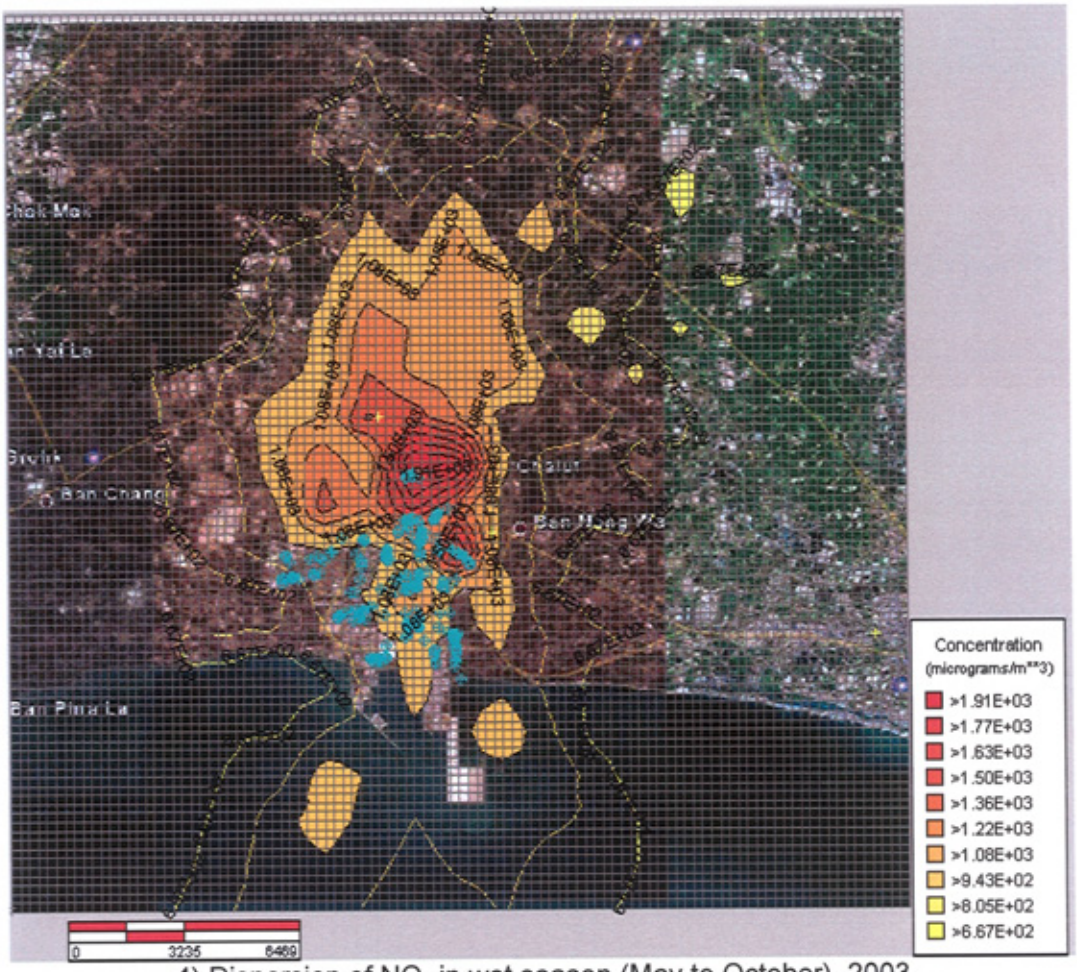
2) Dispersion of NO_x in wet season (May to October), 2002

Figure E.2 ISC model Result of wet and dry season NO_x concentration dispersion from the Maptaphut Industrial Estate (continued).



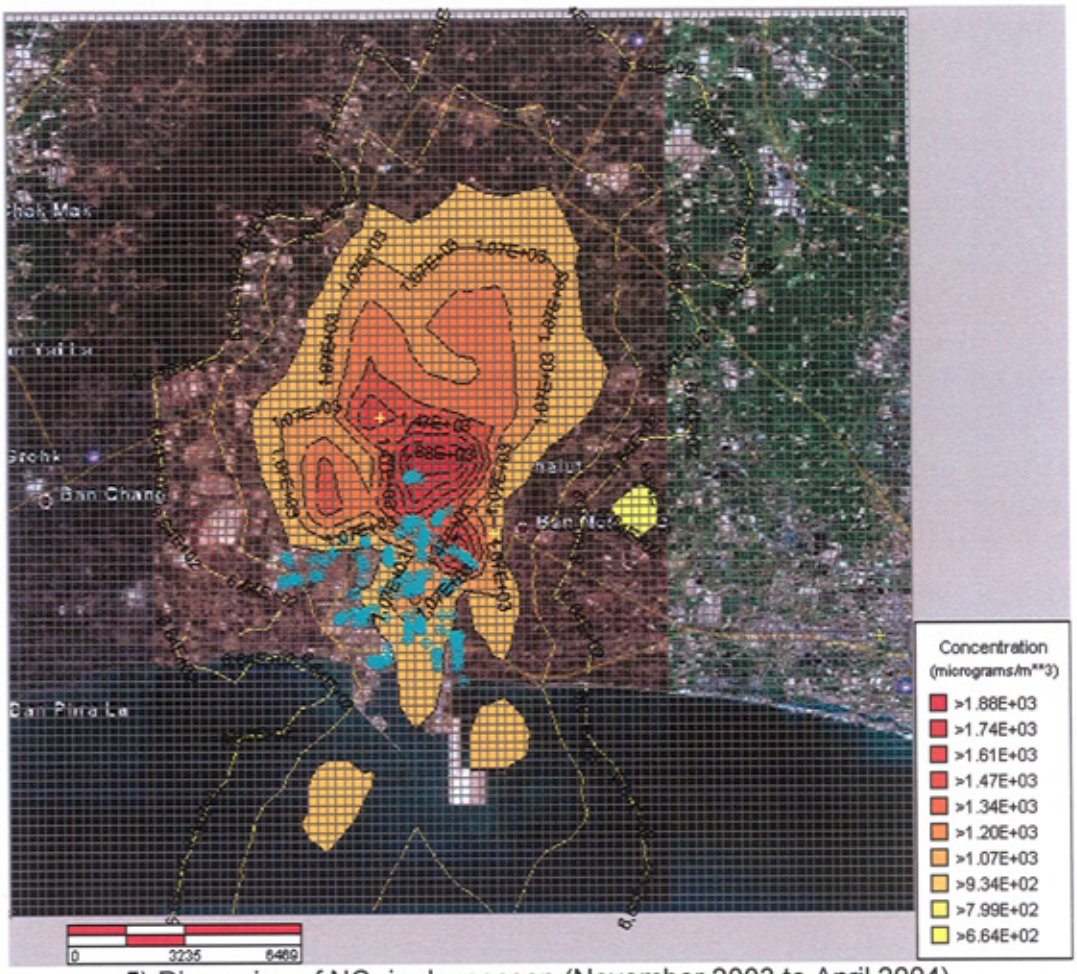
3) Dispersion of NO_x in dry season (November 2002 to April 2003).

Figure E.2 ISC model Result of wet and dry season NO_x concentration dispersion from the Maptaphut Industrial Estate (continued).



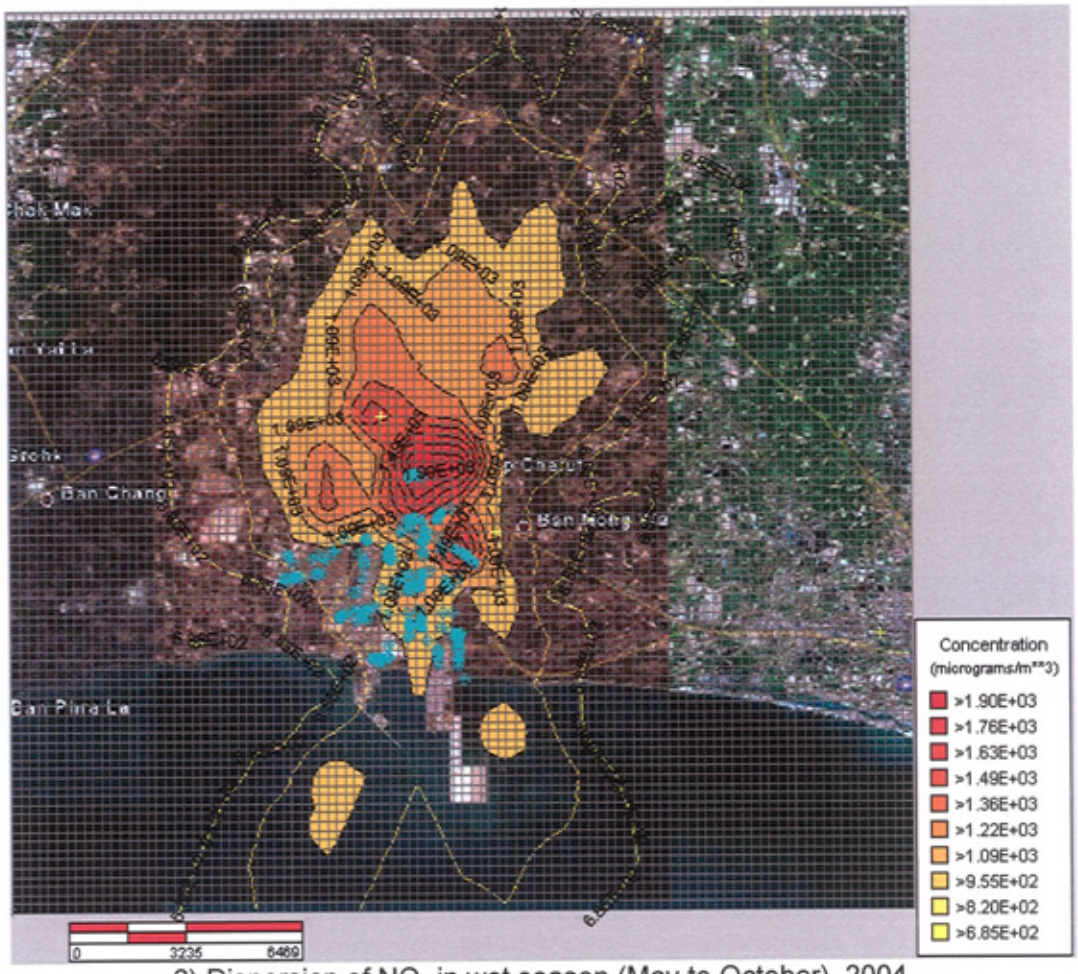
4) Dispersion of NO_x in wet season (May to October), 2003

Figure E.2 ISC model Result of wet and dry season NO_x concentration dispersion from the Maptaphut Industrial Estate (continued).



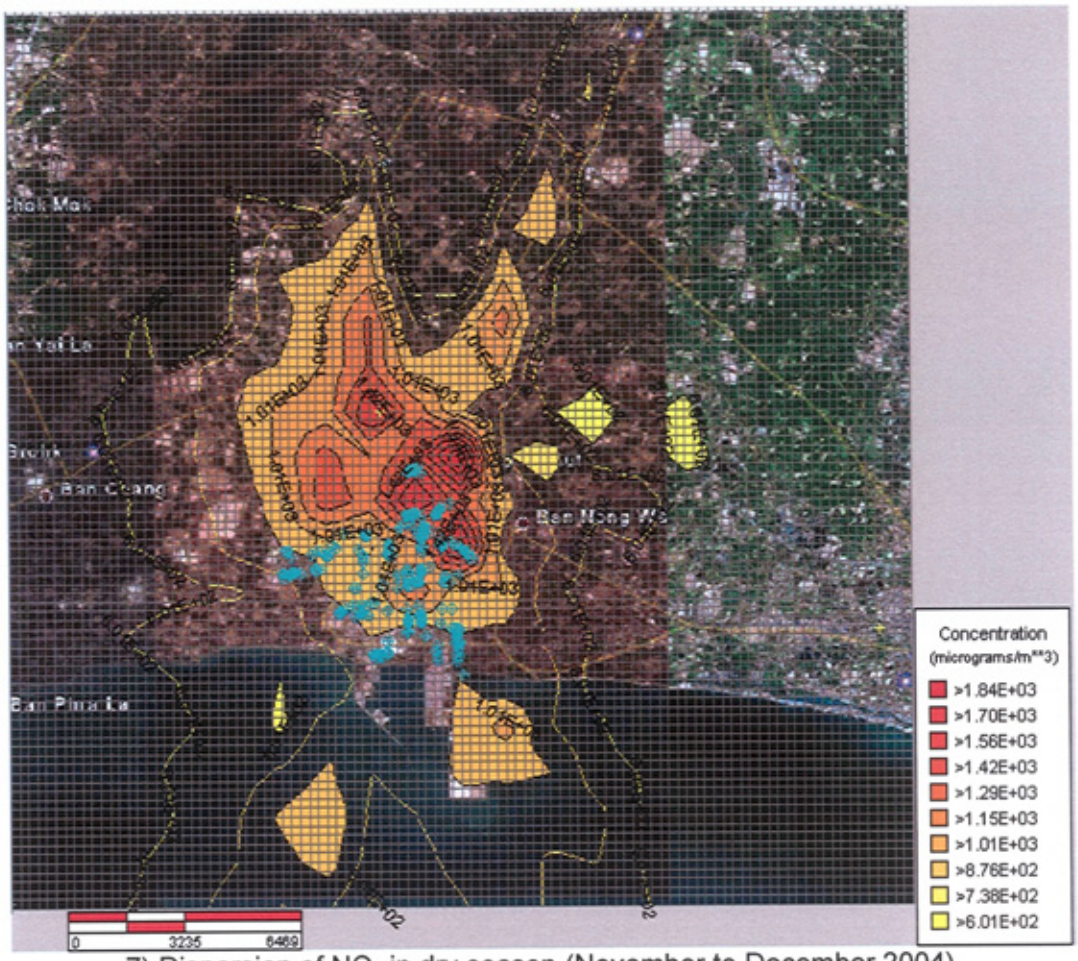
5) Dispersion of NO_x in dry season (November 2003 to April 2004).

Figure E.2 ISC model Result of wet and dry season NO_x concentration dispersion from the Maptaphut Industrial Estate (continued).



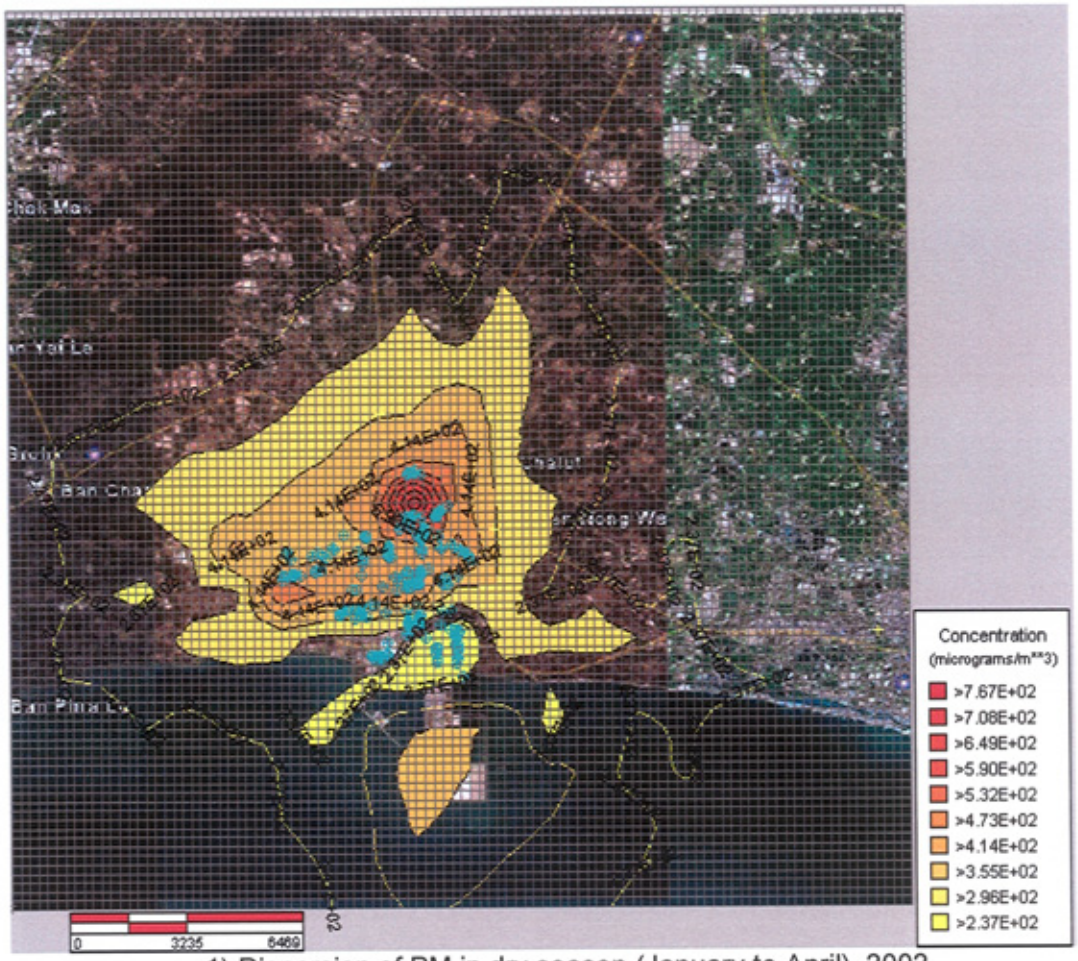
6) Dispersion of NO_x in wet season (May to October), 2004.

Figure E.2 ISC model Result of wet and dry season NO_x concentration dispersion from the Maptaphut Industrial Estate (continued).



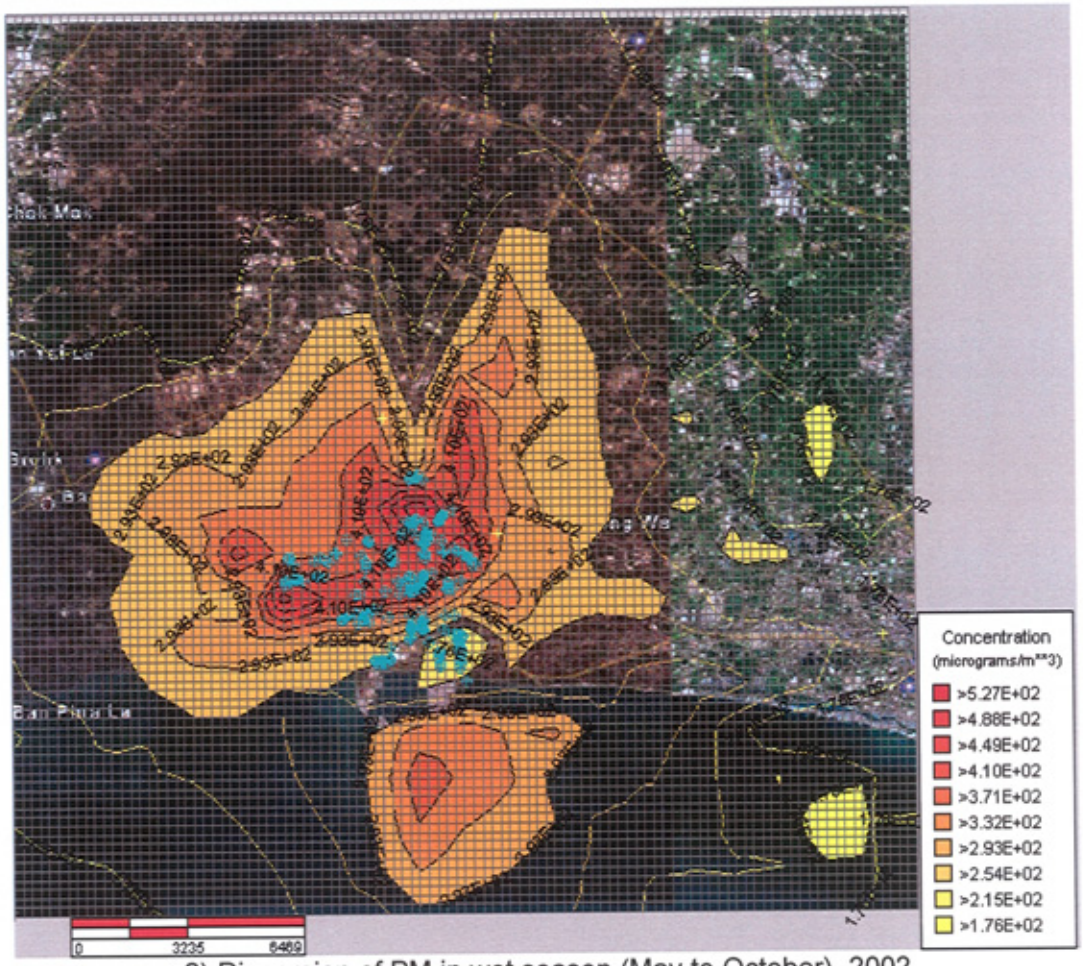
7) Dispersion of NO_x in dry season (November to December 2004).

Figure E.2 ISC model Result of wet and dry season NO_x concentration dispersion from the Maptaphut Industrial Estate (continued).



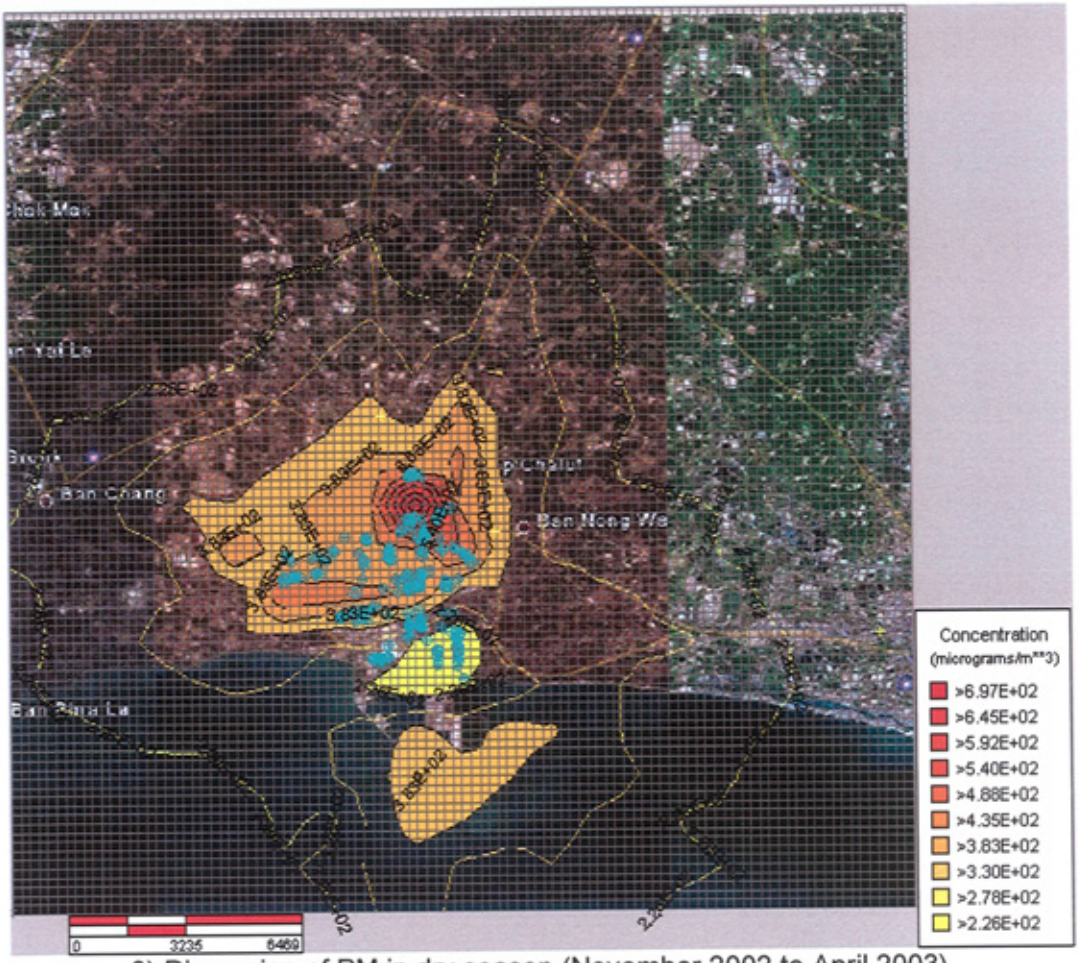
1) Dispersion of PM in dry season (January to April), 2002

Figure E.3 ISC model Result of wet and dry season PM concentration dispersion from the Maptaphut Industrial Estate.



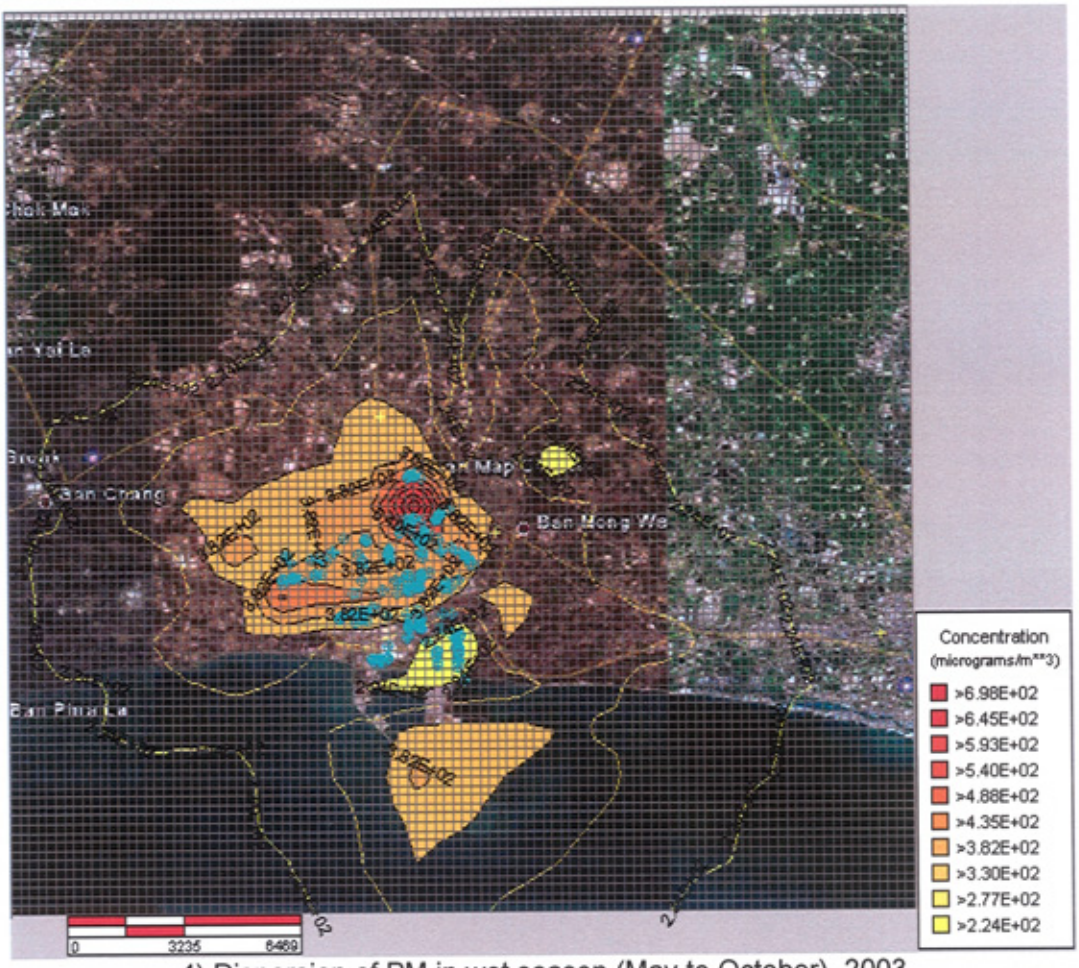
2) Dispersion of PM in wet season (May to October), 2002

Figure E.3 ISC model Result of wet and dry season PM concentration dispersion from the Maptaphut Industrial Estate (continued).



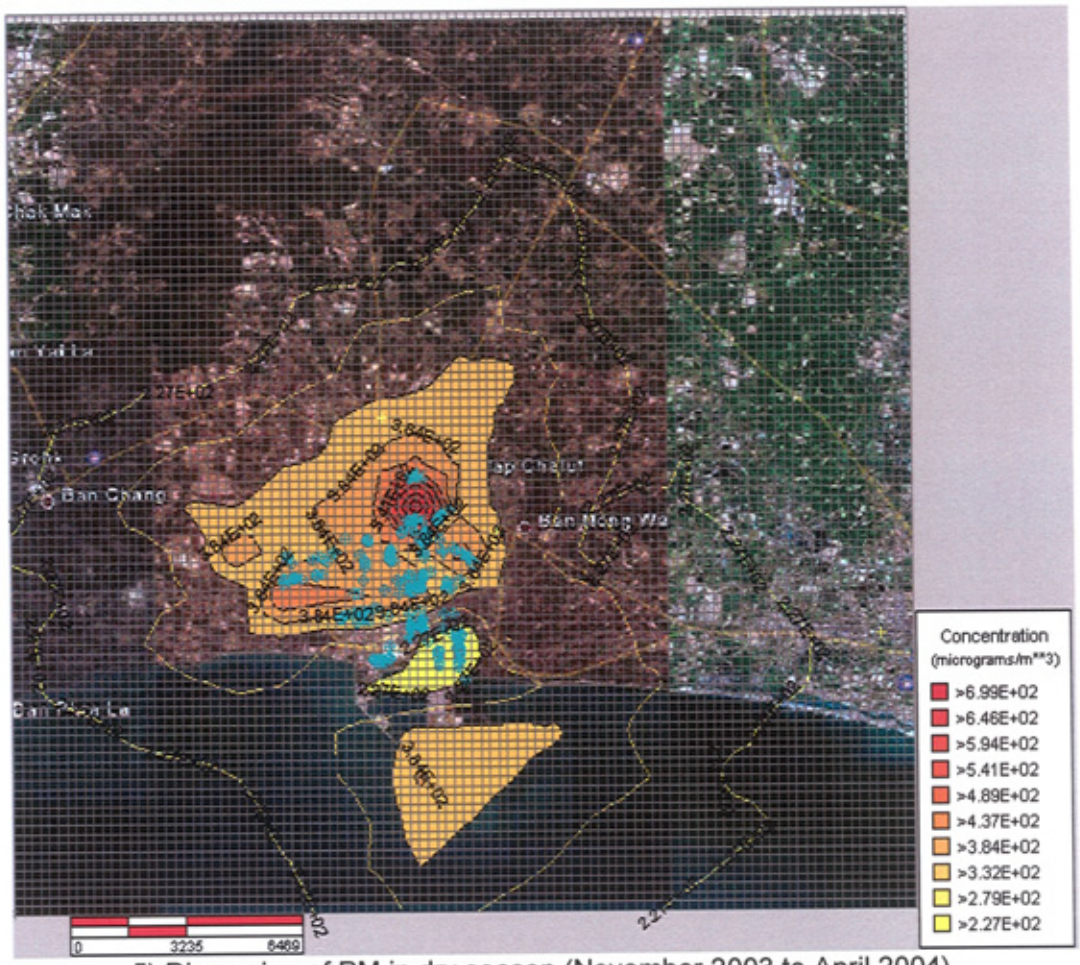
3) Dispersion of PM in dry season (November 2002 to April 2003).

Figure E.3 ISC model Result of wet and dry season PM concentration dispersion from the Maptaphut Industrial Estate (continued).



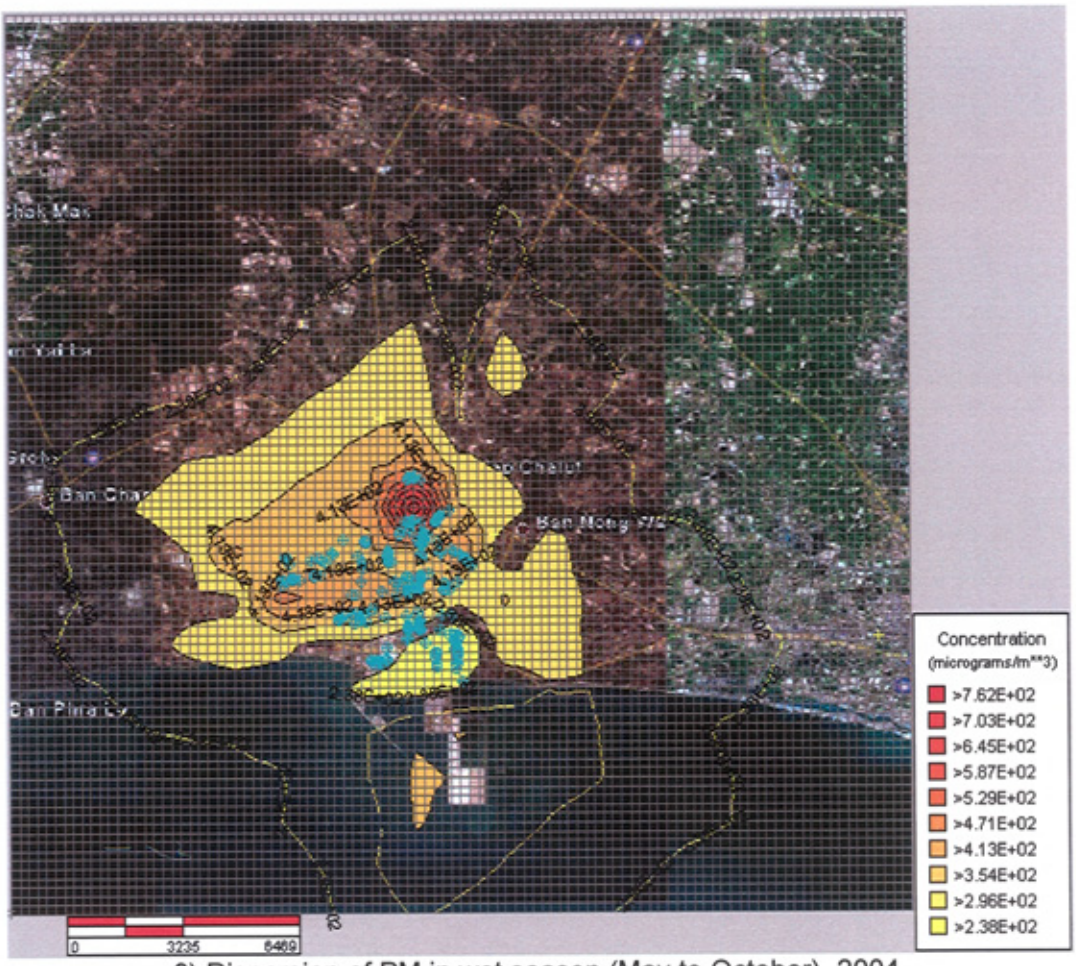
4) Dispersion of PM in wet season (May to October), 2003

Figure E.3 ISC model Result of wet and dry season PM concentration dispersion from the Maptaphut Industrial Estate (continued).



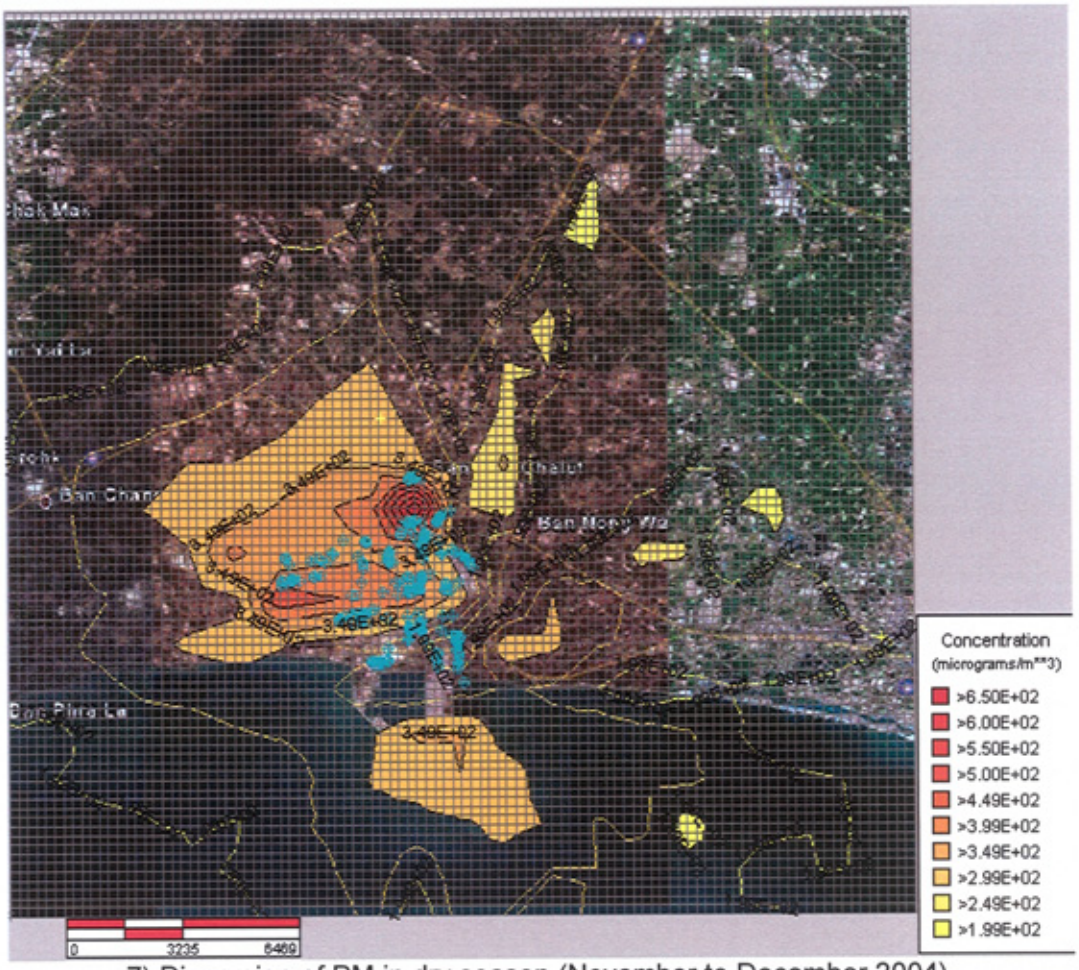
5) Dispersion of PM in dry season (November 2003 to April 2004).

Figure E.3 ISC model Result of wet and dry season PM concentration dispersion from the Maptaphut Industrial Estate (continued).



6) Dispersion of PM in wet season (May to October), 2004.

Figure E.3 ISC model Result of wet and dry season PM concentration dispersion from the Maptaphut Industrial Estate (continued).



7) Dispersion of PM in dry season (November to December 2004).

Figure E.3 ISC model Result of wet and dry season PM concentration dispersion from the Maptaphut Industrial Estate (continued).

Appendix F

Status and Trend of Air Pollution in Bangkok

F1.1 Status and Trend of Air Pollution in Bangkok

The main pollutants in Bangkok from the results of ambient air quality monitoring for more than 10 years are carbon monoxide and Suspended Particulate Matter (SPM), specifically PM_{10}, and CO. The major source is the transport sector. The World Bank's 'Economic Report on Urban Environmental Problems in Thailand in 1993' estimated transport sector emissions of these two air pollutants in 1991 to be 76 and 1,065 thousand tons per year, respectively. It was concluded in the report that the SPM concentration was also indicated as the highest priority air pollution problem in Bangkok (Wangwongwatana, et. al, 2001).

The 1998-2006 ambient air quality data in the general areas and at roadside sites in Bangkok is summarised in Table F1 and Table F2. As indicated, the significant air pollutants along the major roads in Bangkok that exceed the Thai ambient air quality standards are TSP, PM_{10}, and CO. Their concentrations are high enough to result in significant adverse health impacts on the local population.

Table F1 Ambient air quality in general areas of Bangkok in 1998-2006

Nontri Station year	SO2 (ppb) 1 hr avg			NO2 (ppb) 1 hr avg			CO (ppm) 1 hr avg			CO (ppm) 8 hr avg			O3 (ppb) 1 hr avg			PM10(µg/m^3) 24 hr avg			TSP(mg/m^3) 24 hr avg			Pb(µg/m^3) 1 month avg		
	max	avg	min	max	avg	min	max	avg	min	max	avg	min	max	avg	min	max	avg	min	max	avg	min	max	avg	min
1998	136.0	10.3	0.0	122.0	19.3	0.0	8.0	0.9	0.0	4.5	0.9	0.0	136.0	9.7	0.0	164.8	64.6	26.8	-	-	-	-	-	-
1999	120.0	7.7	0.0	118.0	27.9	0.0	8.1	0.8	0.0	3.9	0.8	0.0	109.0	11.0	0.0	177.6	61.7	26.7	-	-	-	-	-	-
2000	151.0	10.8	0.0	130.0	25.0	0.0	5.5	0.6	0.0	4.6	0.6	0.0	127.0	11.4	0.0	145.0	55.8	24.6	-	-	-	-	-	-
2001	43.0	7.4	0.0	101.0	24.7	0.0	6.9	0.8	0.0	4.9	0.8	0.0	104.0	10.7	0.0	108.6	39.7	20.5	0.18	0.09	0.05	0.68	0.16	0.03
2002	98.0	7.2	0.0	140.0	30.9	0.0	6.8	0.8	0.0	4.1	0.8	0.0	128.0	11.2	0.0	56.0	41.7	33.2	0.24	0.09	0.04	0.57	0.12	0.02
2003	52.0	6.1	0.0	129.0	30.1	0.0	3.9	0.6	0.0	2.5	0.6	0.0	134.0	13.0	0.0	173.1	62.5	28.9	0.22	0.10	0.06	0.18	0.08	0.02
2004	68.0	5.9	0.0	95.0	26.3	0.0	7.3	0.6	0.0	4.2	0.6	0.0	115.0	13.2	0.0	183.8	67.5	30.0	0.32	0.12	0.05	0.63	0.10	0.01
2005	37.0	7.1	0.0	99.0	25.3	1.0	4.0	0.8	0.0	3.1	0.8	0.1	118.0	16.6	0.0	107.2	45.5	18.9	0.25	0.11	0.05	0.07	0.02	0.01
2006	43.0	7.4	0.0	123.0	25.7	0.0	4.8	0.8	0.0	3.3	0.8	0.1	136.0	15.4	0.0	111.8	51.0	21.7	0.22	0.11	0.04	0.15	0.07	0.04

Table F2 Ambient air quality at roadside area of Bangkok in 1998-2006

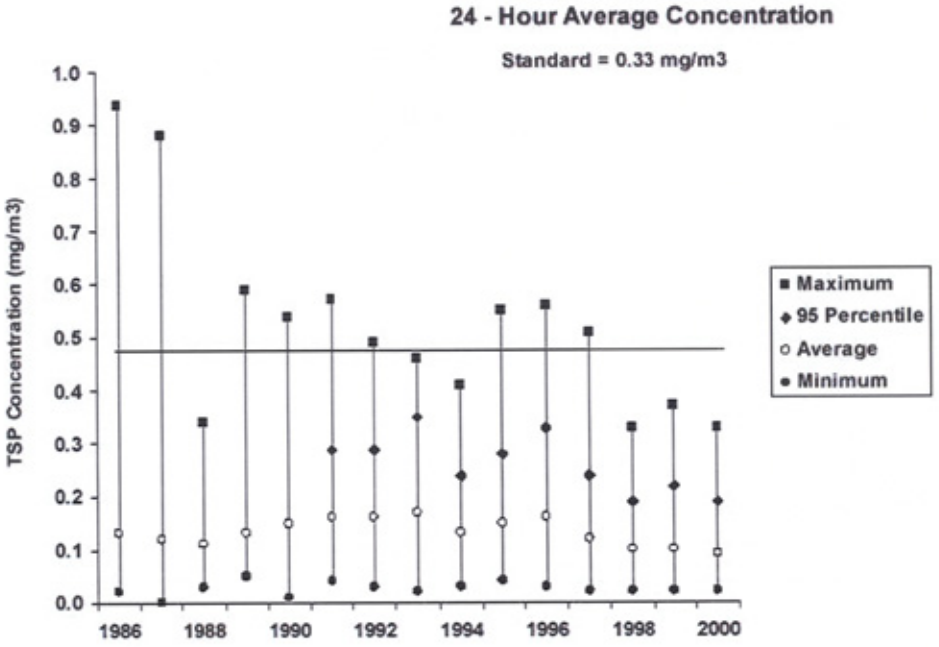
Dindaeng Station year	SO2 (ppb) 1 hr avg			NO2 (ppb) 1 hr avg			CO (ppm) 1 hr avg			CO (ppm) 8 hr avg			O3 (ppb) 1 hr avg			PM10(µg/m^3) 24 hr avg			TSP(mg/m^3) 24 hr avg			Pb(µg/m^3) 1 month avg		
	max	avg	min	max	avg	min	max	avg	min	max	avg	min	max	avg	min	max	avg	min	max	avg	min	max	avg	min
1998	140.0	14.1	0.0	183.0	43.6	4.0	18.7	3.0	0.0	10.0	3.0	0.1	84.0	5.1	0.0	244.5	99.8	46.2	-	-	-	-	-	-
1999	91.0	12.4	0.0	182.0	44.2	2.0	16.1	3.0	0.0	10.7	3.0	0.1	73.0	3.7	0.0	138.6	70.9	34.0	-	-	-	-	-	-
2000	120.0	11.8	0.0	169.0	50.4	0.0	14.0	3.3	0.0	9.6	3.3	0.0	85.0	4.1	0.0	172.5	71.1	27.0	-	-	-	-	-	-
2001	92.0	8.8	0.0	197.0	50.3	0.0	14.4	2.7	0.0	8.8	2.7	0.0	52.0	4.8	0.0	144.5	48.4	23.0	0.35	0.23	0.12	0.3	0.12	0.04
2002	50.0	8.7	0.0	171.0	48.0	0.0	16.6	2.3	0.0	8.5	2.3	0.0	113.0	5.6	0.0	123.1	47.1	25.8	0.37	0.19	0.05	0.43	0.12	0.02
2003	55.0	6.7	0.0	166.0	44.7	0.0	8.8	1.6	0.0	6.1	1.6	0.0	73.0	7.0	0.0	179.9	56.1	17.5	0.33	0.18	0.10	0.25	0.08	0.02
2004	41.0	7.0	0.0	172.0	45.6	0.0	8.0	1.3	0.0	5.7	1.3	0.0	79.0	6.3	0.0	185.1	65.0	29.1	0.52	0.21	0.09	0.35	0.10	0.01
2005	49.0	6.7	0.0	170.0	40.7	0.0	8.6	1.7	0.0	5.6	1.7	0.0	104.0	7.6	0.0	216.0	77.7	13.9	0.45	0.23	0.09	0.23	0.07	0.03
2006	41.0	5.1	0.0	182.0	38.3	0.0	6.9	1.7	0.0	4.5	1.7	0.0	87.0	6.5	0.0	206.2	91.4	29.9	0.40	0.17	0.03	0.11	0.07	0.04

PCD, 2006

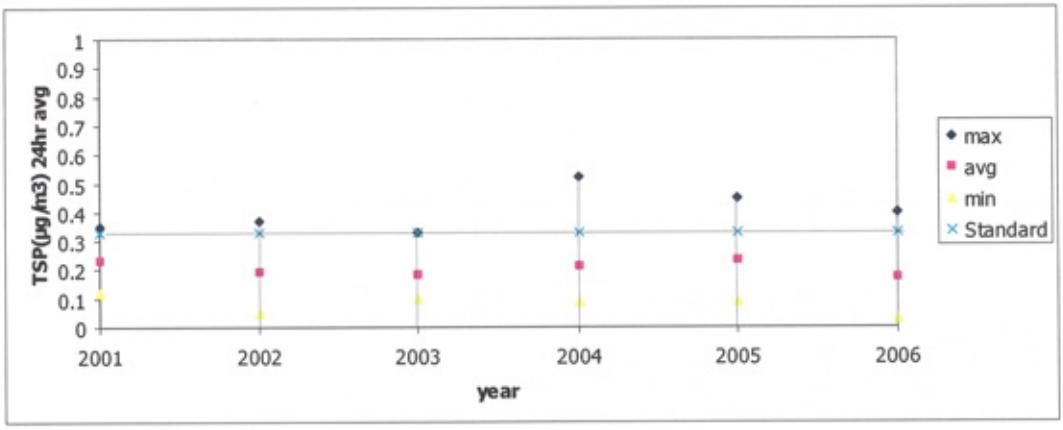
F1.1.1 Total Suspended Particulate Matter (TSP)

Along the streets in Bangkok, because of the congested traffic producing start-stops cycles of vehicles with low speed, the levels of ambient TSP far exceed Thailand's primary ambient air quality standard. In 2000, 24-hour average concentrations of roadside ambient TSP in Bangkok exceeded the standard of 0.33 mg/m^3 at some monitoring sites, with a maximum concentration of 0.48 mg/m^3. The data show that 211 out of 751 observations, or approximately 28 percent of 24-hour average concentrations of roadside ambient TSP exceeded the standards. The annual mean concentration of TSP, 0.19 mg/m^3, also exceeded the 0.1 mg/m^3 standard. The sources of TSP in Bangkok are: diesel vehicles, 40 percent by weight; road dust, 40 percent, and sea salt and particles from industrial sources, 20 percent.

Figure F1 presents the trend of roadside ambient concentrations of TSP in Bangkok from 1986 to 2006. It is obvious that Bangkok has been experiencing high TSP concentrations exceeding the standards every year for more than 10 years. The levels of TSP declined slightly during 1993 and 1994, which is probably due to the construction of the second-stage expressway system that was finished in 1992. In 1993 – 1994, there were fewer numbers of construction projects. In 1995, TSP levels rose again because of the construction of an elevated rapid transit system. In 1999, TSP levels declined again because the constructions in big projects were decrease due to the economic crisis. At present, the roadside maximum concentration of TSP is still higher than Ambient Air Standard.



(a)

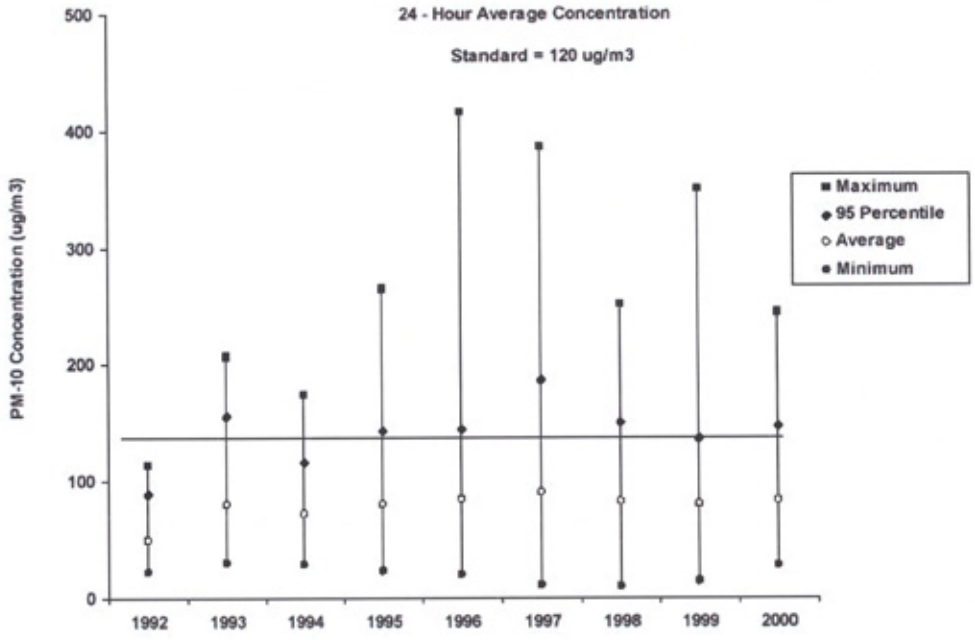


(b)

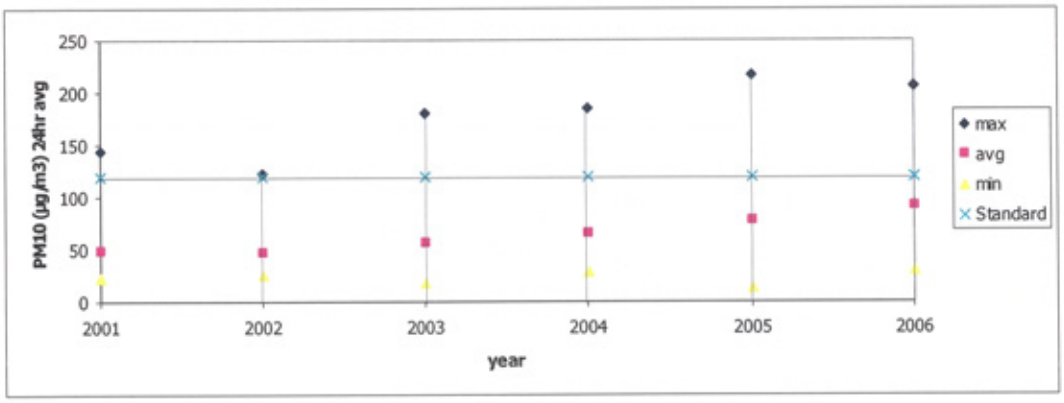
Figure F1 Roadside TSP concentrations in Bangkok, (a) 1986-2000, (b) 2001-2006.
Source: PCD Thailand

F1.1.2 PM10

PM_{10} monitoring in Bangkok began in 1992. From the monitoring data, it was shown that 60 percent by weight of TSP in Bangkok was PM_{10}. In 1993-2006, the roadside ambient PM_{10} 24-hour average concentrations are below Thailand's Ambient Air Quality Standard but the maximum concentration observed exceeds the standard of 120 $\mu g/m^3$. The trend of roadside ambient PM_{10} concentrations in Bangkok from 1992 to 2006 is presented in Figure F2.



(a)



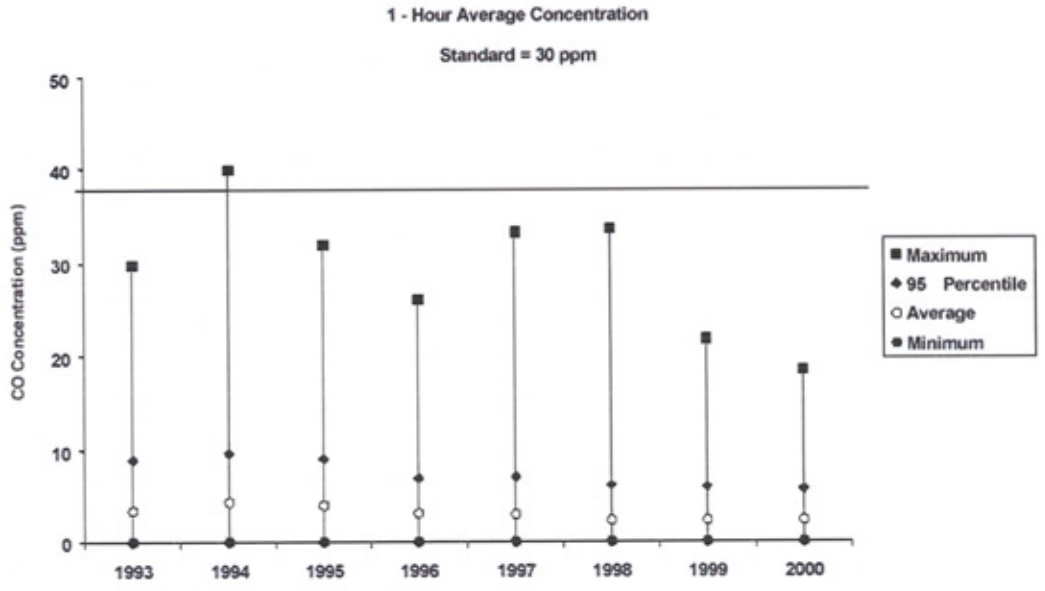
(b)

Figure F2 Roadside PM_{10} concentrations in Bangkok, (a) 1992-2000, (b) 2001-2006.
Source: PCD Thailand

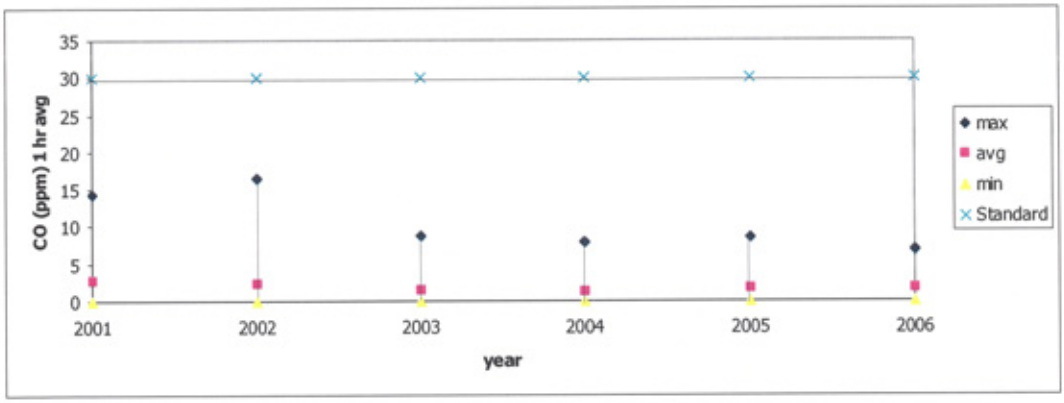
F1.1.3 Carbon monoxide

Roadside ambient concentration of CO in Bangkok was reported as maximum 1-hour and 8-hour average concentrations; only the 8-hour average exceeded the standard of 9 ppm. The effect of traffic congestion in Bangkok was not only in rush hours but also extends throughout most of the day.

Although numbers of vehicles in Bangkok have increased by an average of 30,000 per year since 1990, the roadside concentration of CO has declined since 1993, as shown in Figure F3 and F4.; this has been due to the enforcement of emission standards for new light duty gasoline vehicles since 1992. Vehicles were required to have catalytic converters in order to reduce emissions to meet the standards. In addition, gasoline quality was improved to give oxygen content of between 1-2 percent by weight, thus allowing more complete combustion of gasoline, especially in the existing vehicle fleet.

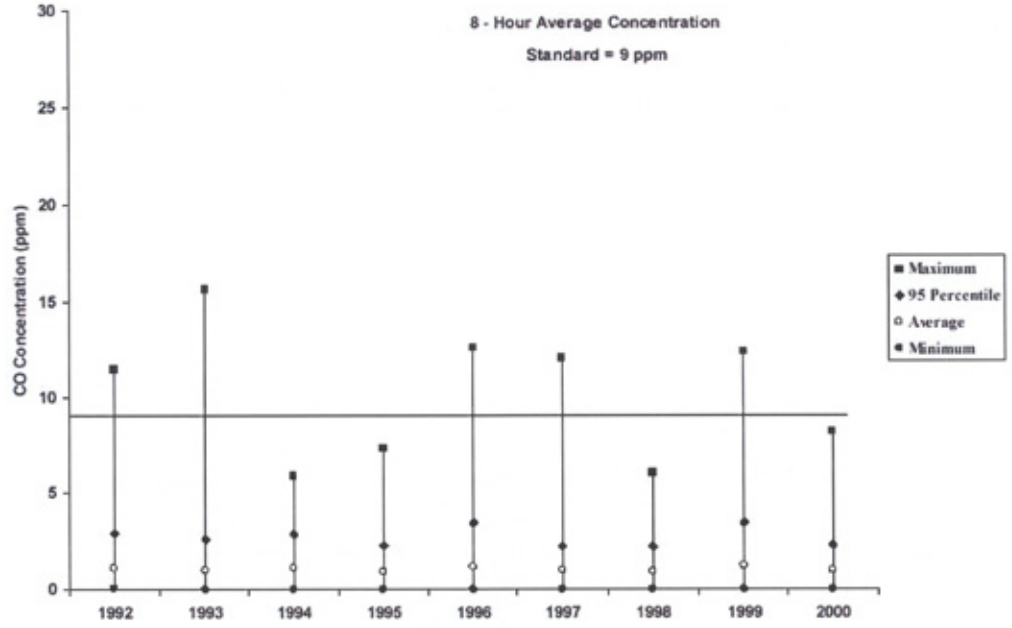


(a)

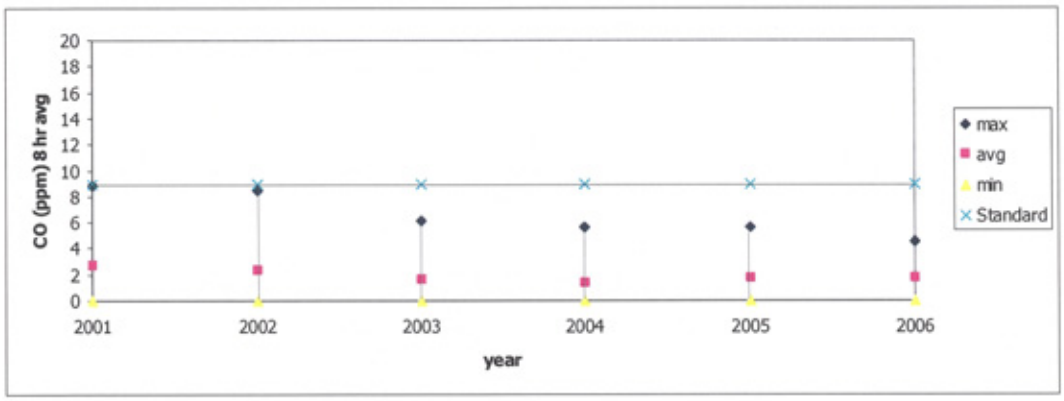


(b)

Figure F3 Roadside CO 1-hr concentrations in Bangkok, (a) 1993-2000, (b) 2001-2006.



(a)

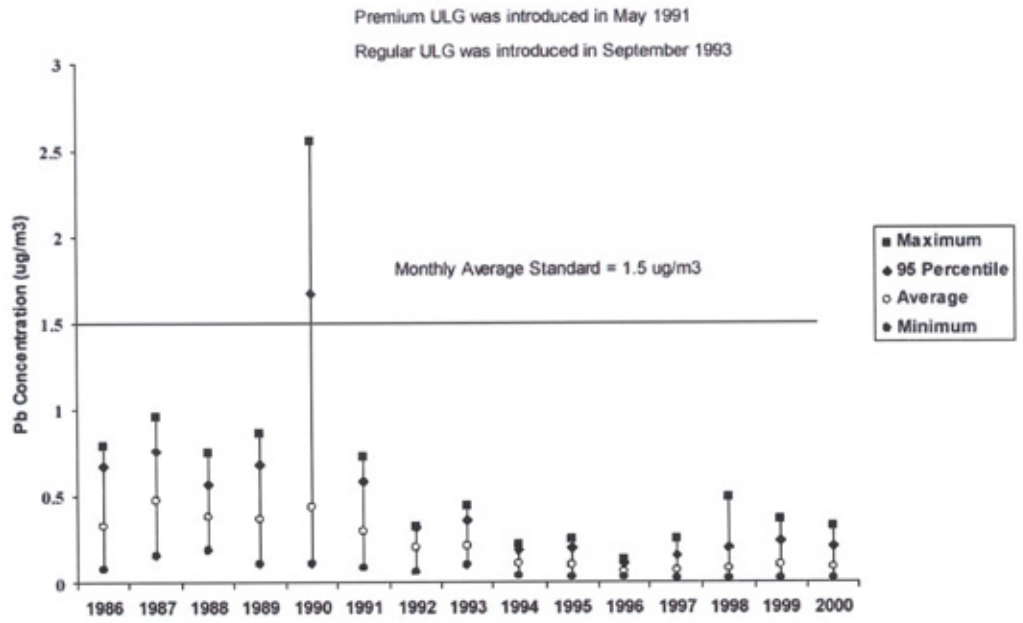


(b)

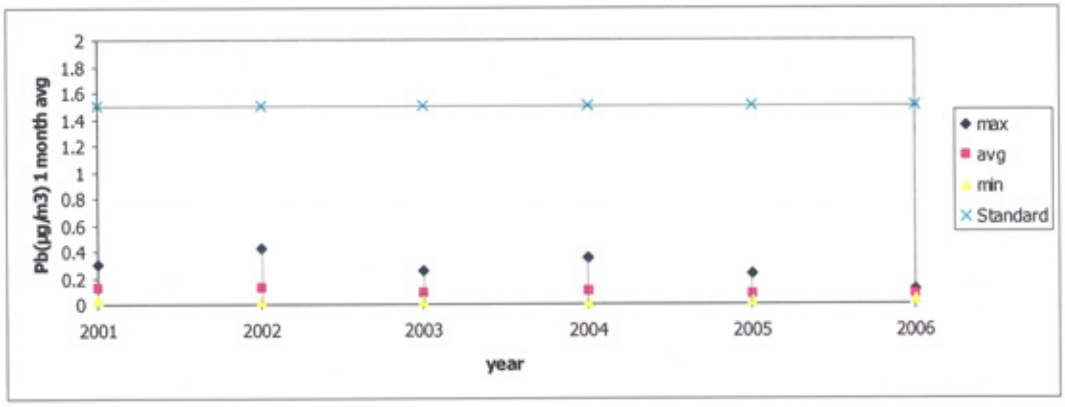
Figure F4 Roadside CO 8-hr concentrations in Bangkok, (a) 1993-2000, (b) 2001-2006.

F1.1.4 Lead

Figure F5 shows the trend of roadside ambient lead concentrations in Bangkok from 1986 to 2006. High concentrations of roadside ambient lead were observed prior to 1990 but started declining progressively since then as a result of the aggressive lead phase-out program introduced in 1990. The current roadside ambient lead concentrations are much lower than the WHO's recommended guideline and Thai Ambient Air Quality Standard.



(a)



(b)

Figure F5 Roadside Pb concentrations in Bangkok, (a) 1986-2000, (b) 2001-2006.

F1.1.5 Other Air Pollutants

Air pollution monitoring data for 1998 to 2006 are hourly ambient O_3, NO_2 and SO_2. These monitoring data are presented in Figures F6-F8. During this time, there is no exceedance of the SO_2 standard in both the general areas and at the roadside areas of Bangkok. Hourly ambient O_3 and NO_2 at roadside are found to exceed the Ambient Air Quality Standard indicative of problems that Bangkok has with photochemical smog; more studies and analyses are required to gain a better understanding of the situation and the associated photochemical reactions.

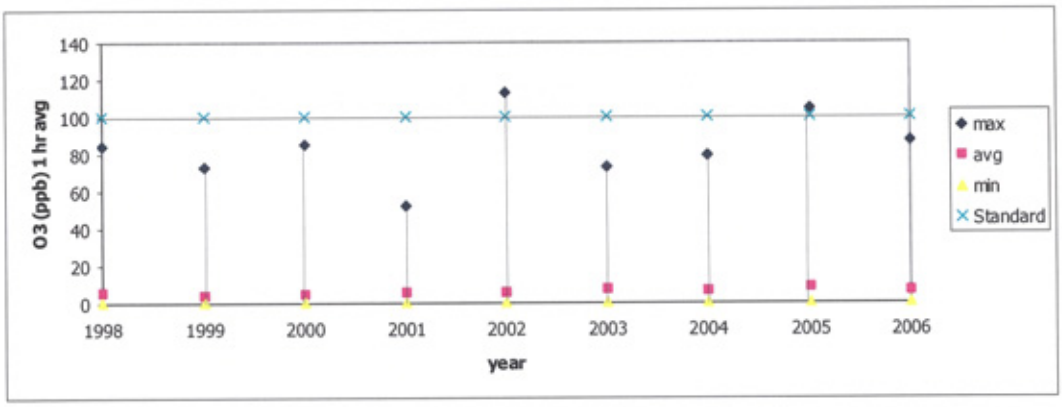


Figure F6 Roadside O_3 concentrations in Bangkok, 1998-2006.

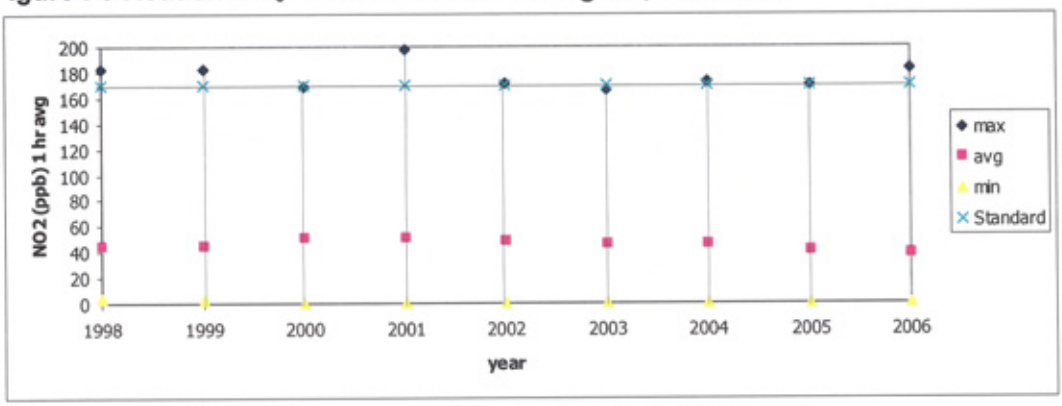


Figure F7 Roadside NO_2 concentrations in Bangkok, 1998-2006.

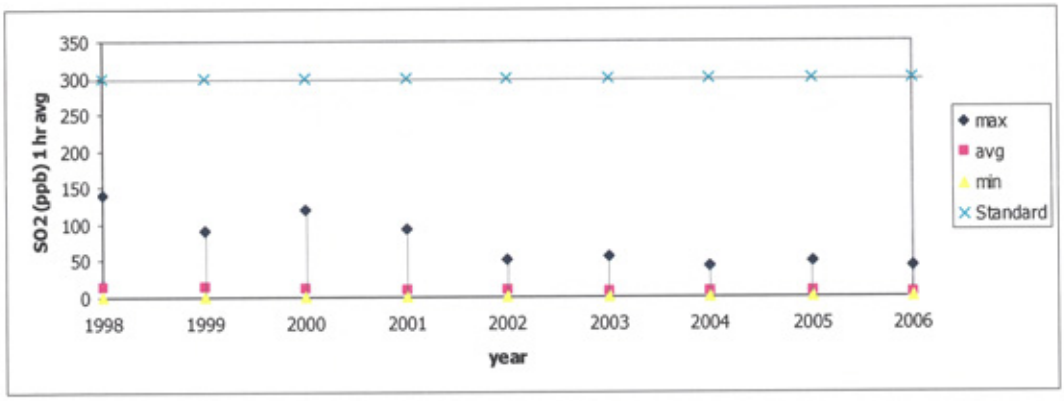


Figure F8 Roadside SO_2 concentrations in Bangkok, 1998-2006.